Marine Payments for Environmental Services in an artisanal fisheries context

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Declaration

I certify that the thesis I have presented for examination for the PhD degree of the London School of Economics and Political Science is solely my own work other than where I have clearly indicated that it is the work of others (in which case the extent of any work carried out jointly by me and any other person is clearly identified in it).

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Abstract

The past decade has seen a growing interest in the application of the Payments for Environmental Services (PES) instrument, in part for its apparent ability to alleviate poverty and inspire sustainable environmental practices. More recently, PES programmes have been advocated for use within marine environments. However, concerns have been raised relating to their applicability in this context, e.g. ill-defined property rights and more fluid environmental services. Yet these issues have received little critical scrutiny.

This thesis presents one of the first empirical analyses of the applicability of PES to the marine and coastal context, more specifically its suitability to small-scale artisanal fisheries.

The first part of the thesis analyses expert opinions in order to identify what opportunities and, indeed, what obstacles remain for PES more broadly in the marine environment. The second part delves a little deeper in order to identify those determinants which can encourage adoption of marine PES within artisanal fishing communities are reported on, paying particular attention to those characteristics important for low-income and vulnerable groups. In addition, the thesis investigates how PES adoption can be influenced by several key design parameters. Analyses are based on primary data collected from six artisanal fishing villages in Mtwara, southern Tanzania.

The thesis presents a number of key findings. Firstly, evidence from expert elicitation suggests that the on-going concerns based on the nature of marine environmental services pertaining to marine PES could be unjustified and solutions for their effective implementation are presented. At the supply-level, fishers' gender and informal risk mitigation strategies are shown to have significant associations with participation within marine PES and may influence the adoption of marine PES programmes within fishing communities. Moreover, whilst PES design can influence adoption, the initial transition away from current management practices can signify a larger utility cost and be met with resistance.

The results have interesting implications for the successful application of marine PES schemes, particularly those hoping to target poor households. The findings are widely applicable due to a global dependence on coastal and marine resources and their continuing degradation.

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Table of contents

List of figures

List of tables

Abbreviations

AES	Agri-Environmental Schemes
ASC	Alternative specific constant
BMU	Beach Management Unit
CB-CCT	Community-Based Conditional Cash Transfer programme
CARE	Care International
CBM	Community-based management initiatives
CE	Choice experiments
CIFOR	Center for International Forestry Research
CLM	Conditional logit model
CPR	Common pool resources
CCTs	Conditional cash transfers
CV	Contingent valuation
CVM	Voluntary Carbon Market
EAME	Eastern African Marine Ecosystem
EPWS	Equitable Payments for Watershed Services project
FAO	Food and Agriculture Organization of the United Nations
HDI	Human Development Index
ICDPs	Integrated conservation and development projects
ICM	Integrated coastal management
IIED	Institute for Environment and Development
ITQs	Individual transferable quotas
JFM	Joint Forestry Management
MBREMP	Mnazi Bay-Ruvuma Estuary Marine Park
MRS	marginal rate of substitution
MRPU	Marine Reserves Park Unit
MPAs	marine protected areas
NBS	National Bureau of Statistics
NGO	Non-government organisation
NLM	Nested logit model
PES	Payments for Environmental Services
PSA	*Pago por Servicios Ambientals*: Costa Rica's national PES programme

REDD	Reducing Emissions from Deforestation and Forest Degradation
SLCP	Chinese Sloping Lands Conversion Programme
SP	Stated preference technique
TASAF	The Tanzanian Social Action Fund
TSh	Tanzanian shillings
TURFs	Territorial user rights for fisheries
UNDP	United Nations Development Programme
VCM	Voluntary carbon market
VLC	Village Liaison Committees
WQT	Water quality trading schemes
WTA	Willingness to accept
WTP	Willingness to pay
WWF	World Wide Fund for Nature

Chapter 1

Introduction

1.1 Payments for Environmental Services and the marine environment

1.1.1 Coastal ecosystems, small scale fisheries and externalities

In a world where almost half of its seven billion population live by the coast, marine ecosystems provide direct and indirect benefits, locally and to a wider global population (Halpern et al., 2012). Mangrove forests, sea grasses and coral reefs provide protection against storms; provide refugia for many juvenile species which later migrate to deeper waters; and are important in sedimentation stabilisation and nutrient recycling. At a global scale, these ecosystems sequester carbon and provide important cultural and tourism areas (Barbier, 2010; Barbier et al., 2008; Nellemann et al., 2009). Together these marine and coastal ecosystems are symbiotic, non-autonomous units in a larger seascape linked by ecological and hydrodynamic processes (Moberg and Rönnbäck, 2003).

Yet, burgeoning coastal populations, intense poverty as well as persistent and destructive fishing practices continue to weaken the health of coastal systems. Fisheries now threaten not only the very resources upon which fishers' livelihoods depend, but also the coastal ecosystem and its capacity to provide beneficial environmental services to these fishers and others, both now and in the future (Berkes et al., 2001; Defeo and Castilla, 2005; Halpern et al., 2012).

Coastal and marine ecosystems[1] are among the most productive ecosystems found on earth. Coastal zones comprise as little as four and eleven percent of the earth's total land and ocean area respectively. Yet, these critical areas represent 90% of all marine fisheries catch and are the source of as much as 61% of total gross world product (Barbier, 2010; MA, 2005; Nobre, 2011; UNEP, 2006). In the mid nineties, offshore gas and oil were estimated have an annual worth of US$ 132 billion, trade and shipping US$ 155 billion, and marine tourism as much as US $161 billion (MA, 2005). More recently, in 2010, capture fisheries and aquaculture were worth some US$ 98.5 billion and US$ 119 billion respectively (FAO, 2012).

[1] Coastal systems are designated as the area which include waters less than 50m deep through to that area inland to a maximum of 100km or 50-meter elevation from the coastline. Marine systems include all those waters extending from the low water mark (50m depth) to the high seas (UNEP, 2006).

Fisheries alone provide a vital source of food, employment, trade and economic well-being for mankind worldwide, in particular to those in low-income countries. Recent estimates place as many as 54.8 million people engaged in capture and aquaculture fisheries, of which over 23 million earn their living from marine capture fisheries (FAO, 2012, 2009, 2005; Lunn and Dearden, 2006). Moreover, the Food and Agriculture Organization of the United Nations (FAO) estimates that a further 200 million people rely on the sector either directly or indirectly for their livelihoods – through ancillary employment in processing, marketing, distribution and equipment processing to name but a few. Together with dependents, primary production and associated employment in the fisheries sector assures the livelihoods of some 660-820 million people or, to put this number into context, 10 to 12% of the world's total population (FAO, 2012).

Small-scale artisanal fisheries are identified as amongst the world's most vulnerable and display a high occurrence of poverty; many still live on the margins of human dignity and 20% are thought to earn less than $1 a day (Béné et al., 2010). As such, these fisheries play a critical role in poverty reduction and food security (Béné et al., 2010; Staples et al., 2004). Not to be underestimated, these artisanal fisheries contribute more than half of marine and inland catch worldwide and comprise approximately 40% of global marine catch destined for final human consumption; moreover, they employ as many as 90% of the world's capture fishers, 95% of which are found within low income countries (FAO, 2010; Lunn and Dearden, 2006).

However, small-scale fisheries are one of the major factors affecting coastal and coral reef health (Defeo and Castilla, 2005; Hawkins and Roberts, 2004). Persistent overfishing and a rising use of destructive fishing gear – in an effort to catch whatever fish remain – results in the untiring and increasing degradation of these areas. In fact, these marine and coastal ecosystems are some of the most heavily exploited; are now considered overfished or collapsed; and continue to deteriorate faster than other ecosystems (Barbier, 2010; Defeo and Castilla, 2005; Halpern et al., 2008; UNEP, 2006; Worm et al., 2009). Globally, 35% of mangrove habitats, a third of coral reefs and approximately 30% of sea grasses are considered either lost or degraded (Barbier, 2010; Hargreaves-Allen et al., 2011).

1.2 PES within the marine environment

1.2.1 Payments for Environmental Services

In areas of prevalent poverty, justifying interventions which serve to reduce fishers effort, catch and ultimately income will prove difficult. Indeed, in the past, many

marine conservation efforts met with high resistance and low compliance for failing to deal with the socioeconomic aspects of many of these fishing communities (Christie, 2004). But compliance and enforcement will not be the only issue, nor should it be: interventions which serve to further marginalise and compound poverty can not be, and should not be, promoted.

The problem here is that those who benefit from the resources are not those paying for its supply. Healthy coastal and ocean ecosystems are important globally, yet these costs are borne locally by coastal communities including those subsistence and low-income fishers.

In the terrestrial context, recent conservation initiatives have shown a growing interest in the use of market-based instruments, in particular Payments for Environmental Services (PES) (Muradian et al., 2013). With a lack of tangible markets preventing resource users from recognising or indeed capturing the benefits of nature's real value, PES schemes are considered a promising new approach for bridging widespread conservation deficits (Engel et al., 2008; Goldman-Benner et al., 2012; Mandel et al., 2009; Tacconi, 2012; Van Hecken and Bastiaensen, 2010; Wunder et al., 2008) as well as addressing many of the inequities associated with the distribution of local conservation costs and more dispersed benefits (Balmford and Whitten, 2003; Engel et al., 2008).

PES create direct markets between service users and service providers. In effect, PES schemes put Coase's theorem into practice, which states that under certain conditions the problems of external effects can, in theory, be overcome via private negotiations between the affected parties (Coase, 1960; Engel et al., 2008). PES move away from the 'Polluter Pays Principle' to a 'Victim Pays Principle', whereby resource owners are no longer seen as the polluters (or the injuring party) but as service providers who can now add one or more environmental services to their production portfolio (Van Hecken and Bastiaensen, 2010). Direct incentive instruments such as PES, it is reasoned, will lead to the more efficient allocation of scarce conservation funds, as well as cover the potentially high opportunity costs associated with protection, particularly in low-income countries (Ferraro & Kiss 2002). As a result, although primarily designed as a conservation tool, many believe PES will contribute to poverty reduction and regional development through the payments made to poor resource users (Engel et al., 2008; Pagiola et al., 2008).

1.2.2 PES and artisanal fishers

Artisanal fisheries are often seen as an activity of 'last resort' and can represent an entry level for policies which target the poor. In the past, marine management tools have largely failed to extinguish unsustainable practices, in part due to an inability to change behaviour, inspire compliance or compensate for loss of earnings; sustainable consumption is the exception rather than the rule (Mohammed, 2012). In the past, marine conservation initiatives have focused on legislative tools such as marine protected areas (MPAs) and individual transferable quotas (ITQs) as well as less direct approaches like integrated conservation and development programmes (ICDPs) (Agardy et al., 2003; Bess and Rallapudi, 2007; Cho, 2005). However, these have seen limited success due to a failing to adequately address the immediate needs of local users; in the worst-case scenarios this has led to the further marginalisation of vulnerable fishers. (Berkes, 2003; Berkes et al., 2001; Davis and Gartside, 2001; Defeo and Castilla, 2005; Pauly, 2006). Perhaps here marine PES can represent a win-win.

In the first instance, can PES contribute to coastal development? To what degree will fishers wish to participate? Who will be those fishers willing to participate? Will marine PES serve to improve the livelihoods of the rural coastal poor or will it in fact serve to exacerbate inequality of these already fragile communities? Given the troublingly low compliance rates within MPA design, will the additional inclusion of an incentive package inspire compliance and behaviour change? How best can schemes be designed in order to promote participation?

Worryingly, in the terrestrial literature, some claim that PES can in fact increase relative poverty in places; under a PES scheme landless poor may become relatively worse off, as profits may be restricted to only those with tenure rights over resources. Barriers to PES may exist based on gender, kinship or other divisions (Landell-Mills and Porras, 2002; Wunder, 2005). Impoverishment is multidimensional and heterogeneous; and as such, interventions can have implications far beyond those initially imagined and factored for. Within artisanal fishing communities, it is clear that these entities are far from homogeneous; fishing communities comprise many strata of 'poor' as well as some who are most definitely not poor by local standards. Marginalisation and exclusion are commonplace (Béné, 2003). On the other hand, well-designed and locally appropriate PES can improve livelihoods and even strengthen local tenure (Evans et al., 2012; Sommerville et al., 2010b) and their voluntary nature may be more socially accepted (Kaczan et al., 2013). If PES schemes are to have a place within small-scale fishery management it is important that their impacts on the welfare of these already impoverished communities are understood, and that they do not further contribute towards the exacerbation of poverty. Enabling

access through instrument design will be fundamental determinants of PES's welfare impacts (Mahanty et al., 2013).

In the second instance, can demand successfully fund such schemes? What are the issues surrounding the realisation of this demand? As yet, few buyers are confident about PES and its potential to deliver the environmental services promised (Wunder, 2005). This problem may be more profound in a marine setting where environmental services are more diffuse, fragmented and to a large extent 'invisible' (Pagiola, 2008). Indeed, how does this underlying difference in environment and human dimensions affect the potential suitability of PES as a marine instrument?

PES is quickly becoming the dominant intervention for biodiversity conservation (Muradian et al., 2013). This recent and rapid interest in PES has unfortunately come with little critical discussion or analysis of the suitability of this approach in the coastal and marine context, or how it can work alongside existing instruments. Coastal and marine ecosystems comprise environmental systems which are both unique and diffuse (Carr et al., 2003; Pagiola, 2008), have ill-defined property rights and are home to some of the most vulnerable socioeconomic groups (Béné 2009). A greater understanding of how these tools can and should transfer to a coastal and marine context and its stakeholders is overdue.

Marine PES may indeed represent a win-win opportunity; however their window for success is tight. Understanding how best to implement these instruments in a complex environment and promoting participation will be key. With many unsuccessful interventions and unfulfilled development promises under their belts, artisanal fishers will require PES to deliver equitably and effectively to permit long-term success.

1.3 Aims and objectives

The growing interest in the application of PES to the marine environment, in part for its apparent ability to alleviate poverty as well as inspire sustainable environmental behaviour, has come with little empirical analysis of these assumptions. As can be seen, questions remain. Who is likely to participate in artisanal marine PES schemes? Are poor and vulnerable fishing communities likely to participate? If not, what restrictions stand in their way? What design schemes are likely to inspire participation? And what challenges remain for marine PES overall, as well as how these can be overcome in order to translate best PES to a marine and coastal environment?

1.3.1 Research aim

The overall aim of this thesis is to explore the feasibility of implementing PES in the marine and coastal context. More specifically, we look at the possibility of using marine PES for coastal protection within artisanal fishing communities.

In the first part of this thesis, we aim to identify what opportunities and, indeed, what obstacles remain for PES in the coastal environment and how best they can be framed within the marine policy portfolio. Taking these findings, possible solutions for implementation are presented.

The second part of this thesis aims to identify determinants which inspire or indeed deter the uptake of marine PES schemes within poor and vulnerable coastal communities. In particular, the role of gender, risk mitigation and informal insurance mechanisms such as social capital and income diversification are investigated. In doing so, the hope is to shed light on barriers which may prevent vulnerable fishers from participating and show that programme design can induce participation. In addition, choice experiments, a stated preference methodology, can provide a good means to test acceptability and adoption rates of PES design. Ultimately, this thesis hopes to improve the design and long-term participation in marine PES.

1.3.2 Research Objectives

The key objectives of this thesis are as follows:

1. Elicit and qualitatively analyse expert opinions on the opportunities and challenges in bringing PES to the marine and coastal environment.
2. Empirically investigate the association of individual characteristics, with particular focus on gender, income diversification and social capital, with the adoption of marine PES schemes by artisanal fishers.
3. Empirically examine fishers' preferences for the design of marine PES schemes and implications for adoption rates.

In order to achieve these key objectives the thesis will also:

4. Produce and analyse primary quantitative and qualitative data which will aid in the successful implementation of PES in the marine and coastal environment.
5. Assess the suitability of the stated preference methodology, Choice Experiments (CE), in determining appropriate PES design characteristics.
6. Produce recommendations for dealing with the challenges in bringing PES to the marine and coastal environment highlighted within this thesis.

1.4 Overview of thesis and methods

The thesis is centred on two distinct but related empirical pieces of research. Both are based on primary data collection.

1.4.1 Expert elicitation

The first part of this thesis relies on primary data collected from experts in the fields of PES and/or marine management. Research data was collated via an online structured survey. Over the period between February and June 2012, 42 expert opinions were elicited and subsequently analysed.

Questions concentrated on the benefits and limitations of implementing PES in a marine setting, the possible use of PES schemes within a wider portfolio of marine conservation instruments and the possible role of PES in coastal development and poverty alleviation. Open-ended responses were coded using a grounded approach.

More details on this methodology are presented in Section 3.4.

1.4.2 Participation choice: Mtwara field research

The second part of this thesis looks at the supply side of marine environmental services through a PES scheme. The first of three chapters synthesises the current thinking on determinants to participation and adoption of new conservation technologies and initiative. It further draws on the fisheries literature to make hypotheses about determinants of participation in a marine PES. The subsequent two research chapters use primary data collected from a field study to determine the possible influence of individual characteristics and programme design on the adoption of a currently hypothetical marine PES scheme. The data for these chapters relies on a case study of six artisanal fishing villages along the coast of the Mtwara Region in Tanzania. The hypothetical marine PES is designed given the most likely restrictions requiring behavioural change given current local circumstances; these are however likely to be common marine PES scheme interventions within the artisanal context.

The region of Mtwara is located in the south of Tanzania and borders Mozambique. The region is considered one of Tanzania's less developed (Malleret, 2004). Research was conducted within Mtwara Region's two coastal districts: Mtwara urban and Mtwara rural. Together these two districts comprise around 26% of the Region's total population: 92,602 and 204,770 respectively (Barr 2010; Guerreiro et al. 2010). Coastal dwellers in these districts exhibit a wide array of livelihoods but show high dependence on fisheries specifically (Malleret, 2004).

Mtwara's coastal waters contain some of Tanzania's most significant biodiversity. Its reef system is of critical importance as a source of larvae and spores to neighbouring regions and is an important area for many mega fauna. In order to manage this important marine and coastal area, Mnazi Bay-Ruvuma Estuary Marine Park (MBREMP) was gazetted in 2000. However, the area continues to allow fishing within its borders and exhibits no restrictions additional to those enforced outside of the park.

Between April and June 2010 fisher surveys were conducted with 662 fishers, 35 were incomplete and dropped. A final useable sample size of 627 was obtained. Both men and women conduct fishing activities in this region. Surveys were conducted with 354 male and 307 female fishers. Survey design followed those guidelines as set out by Bateman et al. (2002), and were shaped from focus groups and key informant interviews. The surveys collected data on individual and household demographics; household assets; attitudes relating to fishing, the environment and conservation; fishing practices and income; diversification strategies of the individual and household and social capital characteristics. A hypothetical scenario was also presented relating to the possible implementation of a marine PES scheme within local waters. Surveys with male fishers also included a CE; the single style of female fishing was not conducive with a CE. Fishers were presented with choice cards which showed two new scenarios alongside the status quo. Options varied over 3 attributes: restrictions on closure area, restrictions relating to allowable net size and size of compensation payment. Surveys were conducted by a team of trained field researchers throughout the months of March and June 2010 inclusive.

More information pertaining to methods, questionnaires and valuation scenarios are provided in relevant chapters. A more detailed review of the case study can be found in Chapter 3.

1.5 Contribution to knowledge

This research makes a number of novel contributions to the research literature on marine PES. To date, little [2] empirical analysis exists which addresses the transferability of PES to the marine and coastal context. Discourse on marine and coastal PES is restricted to policy and discussion pieces. We further provide some of the first quantitative and qualitative data relating to this field, thus adding substantial knowledge to a previously scarce research area.

[2] The author is aware of only one previous empirical paper relating to marine PES: (Barr & Mourato 2009)

A recent interest in marine PES has seen the publication of a number of discussion and policy pieces on the subject. However, the overarching challenges of implementation have received little critical analysis. The first research chapter of this thesis, Chapter 3, is the first to attempt this. The chapter collects and collates primary data in order to analyse what experts in the fields of PES or marine management believe to be the more pertinent issues for successful marine PES implementation. The chapter highlights some of the major elements, giving a sense of magnitude to these issues which have been previously discussed within a wider list of limitations.

Access to PES programmes will fundamentally affect who participates and who does not and ultimately determine what the livelihood implications the scheme will have (Mahanty et al., 2013). Within the PES literature, assumptions about the instrument's ability to target poor members of society continue to be based on assumptions relating to the apparent voluntary nature of the instrument, however other barriers exist. Ex-post studies have examined household factors which enabled participation. To the author's knowledge no ex-ante studies exist which investigate those determinants which may reduce desire to enrol within PES schemes, and to a great extent within overarching development-conservation initiatives overall (Sesabo & Tol 2005). The role of risk mitigation strategies such as social capital and income diversification has received little to no attention, less so within a coastal and marine context where these coping mechanisms have been shown to be important livelihood strategies.

Chapter 6 marks one of the first empirical contributions to the marine PES literature. These results provide an understanding of how gender, income diversification and social capital variables can promote but at the same time dissuade participation in novel marine PES initiatives. To date the author knows of no gender analysis within the PES literature, much less within a marine and coastal PES context. This thesis is one of the first research studies to empirically examine participation and adoption choice of environmental service suppliers within marine and coastal PES schemes. In particular, those determinants of low-income and potentially highly vulnerable individuals are investigated.

PES can further influence adoption rates through its design. While the previous two chapters relate to endogenous individual characteristics, willingness to participate may also be affected by programme design, in particular those restrictions put in place. Previous work has shown that scheme design can influence adoption and re-enrolment rates in agri-environmental schemes (AES). Fishers have also been shown to value management restrictions differently (McClanahan & Mangi 2004).

Chapter 7 is one of the first to demonstrate that CE analysis can be used in the design of marine, and indeed terrestrial, PES instruments. To date, PES design in developing countries has placed little value on this technique (Whittington and Pagiola, 2012). CE is shown to assist in defining how local stakeholders value various restrictions and can highlight preferred options. Economic values are placed upon management restrictions and trade-offs are revealed. In addition, it is shown to be relevant in a low-income small-scale context. Previous work has focused on larger-scale operations and/or public values relating to implementation of marine management options, and overall there has been little application of CE within fisheries management (Wattage et al., 2011). No evidence of the previous use of CE within small-scale artisanal communities was uncovered.

1.6 Thesis structure

The thesis takes the following format:

1.6.1 Chapter 2: Literature review

Chapter 2 provides an overview of the PES literature to date with a particular focus on participation decisions amongst poor communities. This literature is further linked to the current marine and coastal context and more specifically, to small-scale artisanal fisheries.

1.6.2 Chapter 3: Examining the issues in marine PES schemes through expert elicitation

In this first research chapter, determinants of marine PES success are examined. In the first instance the benefits and challenges in bringing PES to a marine environment are explored. This is followed with an investigation into the feasibility of marine PES as a stand-alone tool in the coastal and marine environment and indeed its capacity as a pro-poor mechanism.

In order to answer these questions, Chapter 3 uses a primary data set generated from web-based expert questionnaires. Results from open-ended questions are coded and analysed using a qualitative approach.

We present evidence that the on-going wariness surrounding marine PES could be unjustified and present solutions for their effective implementation. These results are widely applicable due to a global dependence on coastal and marine resources and their continuing degradation.

1.6.3 Chapter 4: Determinants for participation: a review

In this chapter we review the literature surrounding adoption decisions in a number of related and representative fields including conservation agriculture, agroforestry, microfinance and Community-based Management. Household and individual determinants influencing participation decisions are discussed. Where relevant we also look to fishery exit and compliance decisions within the fisheries literature in order to better interpret fisher decision-making processes. The chapter concludes with hypotheses as to how determinants will affect fishers' response to the proposed hypothetical marine PES.

1.6.4 Chapter 5: Site description

Chapter 5 offers a summary of the local context and a description of the case-study site from which the data used in chapters 6 and 7 is collected.

1.6.5 Chapter 6: Determinants of fishers' willingness to adopt a marine Payments for Environmental Service scheme

In the first of the two data chapters, based upon primary data collected from the Mtwara research site, we examine the hypothesis that attributes important in low-income households affect a fisher's decision to join a proposed marine PES scheme. More specifically, we focus on the role that income diversification and social capital can play in this choice. We also investigate the role that gender can play in adoption of marine PES. We do this for two reasons: past adoption studies have been shown to display a gender bias; and fishing communities display strong gender roles. Data on fisher attributes is collected and results are analysed using a logit model.

Evidence is provided showing that participation is significantly associated with a number of social capital variables and income diversification, associations vary between the sexes and show both positive and negative associations. Furthermore female fishers, often the most vulnerable members of society within artisanal fishing communities, are more likely to express a positive decision to participate but are influenced by different facets of social capital to those of their male counterparts.

Results provide an understanding of how gender, income diversification and social capital variables can promote and at the same time dissuade participation in novel marine PES programmes. The chapter goes on to offer some explanations as to possible underlying reasoning behind these results. Results show valuable policy implications for those marine PES schemes hoping to target low-income fishers as well as inferences for improved female participation.

1.6.6 Chapter 7: Investigating fishers' preferences for the design of marine PES schemes.

In addition to individual endogenous determinants, participation in marine PES is anticipated to be influenced by PES design, such as choice and level of PES restrictions.

In Chapter 7 we investigate whether project design influences a fisher's uptake of marine PES programmes. Fishers' preferences for various PES management restrictions were elicited using a choice experiment (CE). Analysis was conducted using conditional and nested logit models. Results indicate fishers' preferences for restriction types and the utility costs associated with each.

The chapter provides an understanding of how various restrictions can influence adoption, and provides insight to other factors which may play equally important roles. The findings present some useful lessons for policy, both in PES design and for the preceding programme foundations.

1.6.7 Chapter 8: Synthesis and conclusions

In this final chapter, the main conclusions of the thesis are presented. Results are discussed with respect to policy implications and recommendations made for the successful application of PES in the marine and coastal environments.

Chapter 2

Payments for environmental services

2.1 Overview of chapter

This chapter provides a summary of the current literature on PES. In particular we cover how the literature relates to a number of key topics. We first provide an overview of PES as a conservation instrument and discuss current levels of demand for environmental service delivery. This is followed by a review of supply dynamics, focusing on the extent to which PES can deliver the concurrent benefit of poverty alleviation it seems to promise. We finish by discussing the current marine management of small-scale artisanal fisheries and the role PES can play in improving conservation results, as well as what implications for poverty may follow.

Further relevant literature is reviewed within each of the chapters.

2.2 Payments for environmental services framework

Underlying the premise of PES, is the failing of markets to ascribe a true value to environmental services and to the free-riding induced by the public-good nature of these services (Van Hecken and Bastiaensen, 2010). More often than not, local monetary benefits of conservation earn less than from alternative uses such as conversion to cropland or non-selective fishing methods. Ecosystem managers, such as farmers and fishers, may be poorly motivated to protect the nature under their guardianship. Payments from downstream beneficiaries can make sustainable resource-use the more attractive option and incentivise the adoption of improved ecosystem management (Engel et al., 2008; Pagiola et al., 2008, 2005).

Although not formally defined in the literature, the most widely accepted interpretation defines PES by the following five criteria: (1) a *voluntary* transaction; (2) PES involve a well-defined *environmental service* (or land use likely to secure that service); (3) the service is 'purchased' by at least one *service buyer*; (4) the service is 'provided' by at least one *service provider*; and (5) the payment is *conditional* on service provision (Engel et al. 2008; Wendland et al. 2010; Wunder 2006).

In reality, PES seldom correspond to this strict definition (Mahanty et al. 2013; Shelley 2011). More recently, the literature has seen a relaxing of the definition with Tacconi (2012) defining PES as "a transparent system for the additional provision of environmental services through conditional payments to voluntary providers".

Sommerville et al. (2009) redefine PES as instruments which 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 (3) additionality and (4) varying institutional contexts. Removed from this definition is the notion that PES must be voluntary. Sommerville et al. (2009) argue that although PES are voluntary at the transaction level – a service provider can decide if to accept payment – they do not necessarily have a choice in the provision of the service (e.g. in such cases where land-use change is illegal).

Although definitions may have relaxed somewhat, conditionality and positive incentives remain at the forefront of PES criterion, and are considered herein as critical facets in their design.

2.3 PES markets: innovative finance

One of their key selling points, PES are touted with the ability to generate additional funding opportunities outside of increasingly constrained government and non-government organisation (NGO) budgets (Balmford and Whitten, 2003; Balmford et al., 2003; Hein et al., 2013; Wunder et al., 2008).

2.3.1 Current market status

Emerging markets are now placing a value on ecosystem services such as carbon sequestration, flood protection and clean water – previously under-valued and over-exploited public goods. And, unlike Overseas Development Assistance and non-for-profit sources, this funding group has grown rapidly in the last two decades.

By their very nature PES are more direct, cost-effective and less institutionally complex than many previous efforts such as ICDPs and regulatory mechanisms; hence are considered more likely to produce the desired outcomes (Frost and Bond, 2008).

Over the last decade, hundreds of PES initiatives have been implemented across the globe (Yang et al., 2013a). For watershed conservation alone, Bennett et al. (2013) catalogued 205 active projects in 2011, with a further 76 in development. Market observers estimated global trade of carbon allowances at US$ 176 billion in the same year; this continues to grow by 11% year on year despite economic volatility (Kossoy and Guigon, 2012). Globally government-mediated payments for watershed protection now exceed US$ 5 billion annually; US markets for wetland and stream mitigation account for transactions worth between US$ 2.4-4 billion each year in transactions, and endangered species mitigation totals US$ 370 million (Madsen et al.,

2011). Global markets for environmental goods and services now reach an estimated US$ 600 billion annually, with projections reaching US$ 800 billion over the next 10 years (Standish, 2006).

In practice, four PES types currently dominate the market (Table 2.1), however to date, most examples of working PES schemes are for the provision of either carbon or water services (Turpie et al., 2008).

Table 2.1 Current PES markets

	Market	*Example of potential PES*	*Buyers*	*Commodity*
1	**Carbon sequestration and storage**	A northern electricity company pays farmers in the tropics to plant and maintain trees	Local, regional and national goverments; international organisations; national carbon funds; conservation groups; land trusts; corporations; hedge funds and investment groups	Assigned-amount units, certified emission reductions, emission-reduction units, carbon offsets/credits, tradable development rights, conservation easements
2	**Watershed protection**	Downstream water users pay upstream farmers to adopt land uses that limit deforestation, soil erosion, flooding risks, etc	Municipalities; private water suppliers; public water suppliers; bottled water companies; farming organisations; hydroelectric energy providers	Watershed management contracts, water quality credits, water rights, land acquisition/lease, salinity credits, transpiration credits, conservation easements, certified watershed-friendly products, stream-flow-reduction licenses, reforestation contracts, protected areas
3	**Biodiversity protection**	Conservation donors pay local people to set aside or naturally restore areas to create a biological corridor	International and national NGOs; private businesses (offsets)	Protected areas, bioprospecting rights, biodiversity-friendly products, biodiversity company shares, debt-for-nature swaps, biodiversity credits, conservation concession, land acquisition, biodiversity-management contracts, logging rights acquisition, tradable development rights, conservation easements
4	**Landscape beauty**	A tourism operator pays a local community not to hunt in a forest in which tourists view wildlife	International and national NGOs; private tourism operators;	Entrance rights, long-term-access permits, package-tourism services, natural-resource management agreements, ecotourism concessions, photographic permits, land acquisition, land lease

Adapted from (FAO, 2007; Landell-Mills, 2002; Wunder, 2006)

2.3.2 *To what extent will PES instruments prove cost-effective and efficient?*

PES are touted as more cost-effective and efficient than other less direct policy interventions; this is perhaps their key selling point (Ferraro and Kiss, 2002; Groom and Palmer, 2010). This thinking is based on a number of assumptions within the PES model, most of which are rarely met (Delacote et al., 2013; Engel et al., 2008; Wunder et al., 2008). Effectiveness requires that PES schemes lead to an increase in environmental services compared to the "business as usual" alternatives. Efficiency pertains to a maximisation of the environmental services obtained from a given budget (Engel and Palmer, 2008). PES have a lot to live up to, and in reality have a long way to go in order to match these claims. The extent to which PES schemes can meet the promise of both of these criteria is hotly debated within the academic field and continues to drive research and contract redesign.

The effectiveness of a PES instrument is determined by targeting, additionality, permanence, leakage and fairness (Engel et al., 2008; Narloch et al., 2013; Pattanayak et al., 2010). But getting these right requires a more in-depth understanding of the local context and ultimately higher transaction costs. The transaction costs will, for a large part, determine the efficiency of a PES programme. Ultimately PES schemes are a trade-off between the two, and the increased transactions costs may outweigh the benefits from improved targeting and pricing (Engel et al., 2008; Ferraro, 2008; Ferraro and Kiss, 2002; Vatn, 2010). Indeed, when transaction costs are high, Coase's theorem calls for a switch to regulatory instruments (Coase, 1960).

In general, transaction costs are highest when projects involve many smallholders and multiple PES actors, where institutions and property rights are weak, and when costs of gaining baseline data, monitoring and enforcement are high (Jack et al., 2008; Wunder, 2007): characteristics which are common in a developing country context. A review of carbon-sequestration schemes within various developing countries reported transaction costs as high as 45% of total costs (Cacho et al., 2005).

2.3.2.1 Targeting

PES programmes start with the decision of who to target for payment. Who should be paid for service provision often depends on the scheme's objectives (Alpízar et al., 2013). This choice however will affect effectiveness as well as efficiency. A more effective PES will target those who can provide the environmental service at the lowest cost. However doing so requires identification of these individuals, not always an easy task given issues of changing baselines and information asymmetry, to name a few (Ferraro, 2008).

Recent publications in the field have suggested a number of methods to better target and implement PES (Wendland et al., 2009). Location is a key consideration; the geographical targeting of high threat/low opportunity cost lands is a win-win for PES and is already implemented to maximise the cost-effectiveness in many protected area and reserve designs (Gauvin et al., 2010; Naidoo et al., 2006; Newburn et al., 2005; Wendland et al., 2009). Practices which protect biodiversity in proximity to protected areas, for example, can buffer impacts on these areas and therefore have benefits beyond those associated with the land change itself. Payments that disregard the degree of risk to the environmental service will also lead to inefficiencies. Failure to take these differences into consideration will result in over-payment for less desirable or low risk areas, as well as under-payment for, or exclusion of desirable land (Alix-Garcia et al., 2005; Pagiola et al., 2004).

Efficient payments will also tend to favour larger resource users with economies of scale and generally not those with smaller or marginal holdings. However, incentive schemes which ignore and/or further marginalise poor resource owners are hard to get behind; indeed the long-term sustainability of PES programmes has been linked to perceptions of equity and legitimacy (Narloch et al., 2013). Many PES now focus on co-benefits such as poverty alleviation, livelihood protection and regional development (Bulte et al., 2008). Moreover, failure to pay resource owners who have been providing environmental services in the past, while paying others to adopt provision has been shown to lead to inefficiencies and moral hazard (Alpízar et al., 2013).

The simplest of payments, is a 'one price fits all' where resource owners are paid for the quantity of land or a change in practice, irrespective of the quality and/or marginal benefit of said land. However, where costs and benefits of biodiversity conservation differ spatially the cost-effectiveness of these uniform payments is found to be low (Wätzold and Drechsler, 2005). More complex PES schemes look towards price differentiation to improve PES efficiency but in the process face increased transaction costs (Engel et al., 2008; Ferraro, 2008). Cost targeting enables prices to more readily reflect the true opportunity cost of service provision and would enable inclusion of a greater area overall for a given budget (Ferraro, 2008; Wunder et al., 2008). Cost targeting can further be designed to include other factors such as potential gains and future risks in addition to cost (Wünscher et al., 2008).

Calculating appropriate payments requires an accurate calculation of suppliers' opportunity costs. The challenge is to identify contract prices in the absence of any market where service providers have little incentive to reveal their true costs (Ferraro,

2008). Transferring price estimates between projects is difficult since implementing new environmentally sound practices can be highly location- and activity-specific (Jindal et al., 2013).

To reduce informational rents to resource owners, conservation agents can take one of three approaches: 1) acquire information on observable landowner attributes that are correlated with compliance costs; 2) offer landowners a menu of screening contracts; and 3) allocate contracts through procurement auctions (Ferraro, 2008). Perhaps the most effective of these is procurement auctions. Unlike screening contracts, auctions do not require that conservation agents specify landowner types and/or quantity of environmental service provision. Auctions invite landowners to competitively bid for contracts against other service providers, and in doing so reduce the incentive for sellers to inflate their costs (Ferraro, 2008). Auctions have a number of advantages over other methods: differentiated payments have been associated with improved additionality (Newton et al., 2012); auctions are also adaptable over time – as conditions change so too will opportunity costs and repeat auctions can keep up with these dynamic scenarios; and more recently, auctions have seen successful implementation in a developing country context (Jindal et al., 2013). However, auction mechanisms require a large pool of bidders in order to induce competitiveness and prevent collusion and can induce self-selection related to economies of scale (Jack et al., 2008a).

To further overcome high transaction and implementation costs, some PES programmes look towards 'bundling' (Wendland et al., 2009). 'Bundling' refers to the selling of environmental services as joint products. This can be beneficial for those currently interested in protecting services which generally do not receive as much attention as those such as carbon. Furthermore, services for which it is often harder to mobilise funding and which at a lower volume may be subjected to higher transaction costs (Landell-Mills and Porras, 2002; Robertson and Wunder, 2005; Wendland et al., 2009).

2.3.2.2 Monitoring and enforcement

The success of a PES market will depend on its ability to monitor and enforce service delivery, and do so at a reasonable cost (Landell-Mills, 2002).

Within PES, payments are expected to be contingent upon the continuous production of environmental services and as such should be linked to clear environmental service monitoring. In reality, PES projects rarely include explicit frameworks for monitoring and evaluating their success and, as such, calculating additionality remains difficult

(Wunder, 2007). In fact, for many PES initiatives worldwide: results are either loosely monitored or not monitored at all; payments are made upfront and in good faith; and are not continuously or truly contingent upon service provisions (Robertson and Wunder, 2005; Wunder, 2007). Monitoring has been identified as one of the hardest criterion to meet. The lack of low-cost monitoring options represents a large barrier for PES effectiveness (Kroeger and Casey, 2007); however, a focus on monitoring within the field and advances in technology are likely to ease this difficulty in the future (Alston et al., 2013; Sommerville et al., 2011).

This said, greater emphasis on monitoring might not adequately solve the problem of assessing additionality. The complexities within systems mean that, in practice, PES schemes will often rely on observable proxies such as actions or outcomes as a measure of success, because direct monitoring is either near impossible and/or too costly (Jack et al., 2008). Performance-based payments are bound to the outcome of a desired environmental good or service. Action-based payments, on the other hand, pay for a pre-defined action or measure (Derissen and Quaas, 2013). The most typical solution is not to pay for the environmental service itself but the land uses which are hospitable to biodiversity (Engel et al., 2008; Pagiola et al., 2004). However, what is often thought to provide a service, such as hydrological services, is often not based upon sound scientific evidence but built upon perceived rather than factual linkages (Wunder, 2007). The future efficiency of PES schemes may therefore rely upon the correlation between these proxies and the provision of the environmental good. Output-based monitoring , can be based on threats, changes to species or presence of rare indicators, to name but a few, and no one criteria will transfer to all PES (Sommerville et al., 2011). However, even for those services that are becoming more easily measurable due to scientific and technological advances, establishing a credible counterfactual remains difficult (Alston et al., 2013).

What monitoring does provide, at its very least, is the promotion of compliance and this is paramount to the success of any PES intervention. PES has been championed as a means to move away from previous, and often ineffective, regulatory mechanisms; the market will facilitate compliance, for example. This assumption, however, relies on the existence and effectiveness of strong institutions (Miteva et al., 2012). In reality, it is unlikely that PES will work without adequate levels of monitoring, enforcement and the appropriate sanctions (Barbier and Tesfaw, 2012; Chhatre and Agrawal, 2008; Engel et al., 2008; Gibson et al., 2005). Enforcement lowers the marginal value of non-compliance and as a result increases the level of compliance as individuals find it more economical to reallocate their time to alternative activities (Muller and Albers, 2004; Robinson et al., 2012). The flipside is that the provision of

enforcement officers can incentivise PES participation through curtailing expected returns from previously unsustainable practices (Chhatre and Agrawal, 2008; Yang et al., 2013a). In some instances, the fear of being caught has proved a greater deterrent to non-compliance than the PES payment itself (Sommerville et al., 2010b; Travers et al., 2011). However, as with previous command and control policy tools, enforcement can prove expensive, particularly when contracting with many stakeholders (Grieg-Gran et al., 2005). Moreover, where labour and resource markets are missing, as they are in low-income and rural settings, enforcement will induce less conservation per dollar as alternatives activities may not offer high enough rewards (Muller and Albers, 2004).

This said, evidence shows that the cost of enforcement and levels of non-compliance can be reduced with good design. Performance-based payments can reduce the need for enforcement but will still require high-levels of monitoring (Derissen and Quaas, 2013; Engel et al., 2008; Ferraro, 2001; Kroeger and Casey, 2007). More recently, inclusion of certain criteria in contract design, such as revealed bidding, has been shown to promote high rates of self-enforcement (Jindal et al., 2013). Where PES requires provision by many, community contacts which promote social norms can further improve compliance (Chen et al., 2009; Clements et al., 2010). Involving environmental service suppliers within monitoring and enforcement decisions empowers PES participants and can enhance long-term protection (Barbier and Tesfaw, 2012). A recent study by Travers et al. (2011) indicates the importance of self-organisation including the ability to devise, monitor and enforce a set of rules. Under a common pool resource game played in four Cambodian villages, those treatments where self-organisation was promoted showed the greatest effects on reducing individual extraction. Community monitoring has been shown to be a cost-effective monitoring and enforcement tool in 'Reducing Emissions from Deforestation and Forest Degradation' (REDD+) programmes (Danielsen et al., 2011; Larrazábal et al., 2012). However, the effectiveness of community monitoring and enforcement may be limited by group size, with the enforcement of larger groups proving more difficult (Poteete and Ostrom, 2004; Yang et al., 2013b)

More generally, PES can increase local incentives to self-enforce thereby reducing required programme or state-level enforcement (Engel and Palmer, 2008).

2.3.2.3 Additionality

Initial results suggest that the potential of PES schemes to increase the conservation of environmental services has been mixed, both between projects and between analyses of the same programmes.

In Costa Rica's national PES programme *Pago por Servicios Ambientals'* (PSA) – whereby farmers were paid to maintain land under forest cover – evidence suggests low additionality. An estimated 76.8% of forest area under the PSA would have been conserved or managed with limited extraction without the PES intervention. In addition, 70% of PSA forest protection contracts were on land that had production capacities limited to forest management/protection (51%) or severely limited agriculture (20%) (Wünscher et al., 2008). Pfaff et al. (2008) further indicate that annually as little as 0.08% of those forests under PSA contracts would have been cleared if payments had not been received. For Mexico's similar national PES programme calculated average clearage rates without the programme were also low: 0.8% per year (Alix-Garcia et al., 2012). These results are even more profound when one considers the baseline. In the example of Costa Rica, a static baseline was employed to define payments; in reality, the true baseline was one where national forest cover was increasing. As such this PES has paid above and beyond what was required and in addition has paid for forest establishment which would have occurred irrespective of payments (Wunder, 2007). Conversely, current CDM criteria employs a rigid adherence to the static baseline ignoring the argument for one which is declining, as posed by many who see natural resource use as an integral part of any resource rich countries' development progress (Wunder, 2007).

The practical reality is that measuring PES success is difficult and requires comparison with a 'business as usual' counterfactual. However, in most instances, PES projects do not construct realistic counterfactual scenarios which consider what could hypothetically occur to the environmental service in the absence of the scheme, nor are payments targeted based on this information (Wunder et al., 2008).

For those PES schemes, which can claim additionality, further possible concern relating to permanence and leakage can affect true levels of additionality.

2.3.2.4 Leakage and Permanence

Leakage is associated with the inadvertent relocation of activities which become restricted under a PES programme (Engel et al., 2008). Leakage can occur for two reasons (Alston et al., 2013). First, PES can lead to the displacement of environmentally damaging activities to an area outside the target zone. Secondly, leakage can occur due to raising commodity prices for goods restricted under the scheme, whereby non-participants locally, or not, are motivated to carry out the activity thereby generating the environmental externalities anyhow (Alston et al., 2013; Murray and Sohngen, 2004; Wunder et al., 2008).

Leakage is only a relevant concern where the spatial scope of the intervention is lower than that of the desired service. And as such leakage will always be a consideration for global services such as carbon and biodiversity markets (Wunder et al., 2008). However, programmes can be more successful in reducing leakage when they are larger in scale (Alston et al., 2013). As such, the contracting of community property rights can offer economies of scale and reduce possible issues of leakage, although may ultimately increase transaction costs and so success will depend on the level of community institutions in place, the strength of collective action and/or effective enforcement.

In practice little is known about leakage, mainly due to the difficulty in accurate calculation. Of 14 PES-like programmes analysed by Wunder et al (2008) only one quantitatively attempted to estimate project leakage, finding potential estimates varying from small to an upper bound 21%.

Permanence, on the other hand, refers to the ability of PES schemes to achieve long-term improvements in environmental service provision, including the time-period after which payments are stopped (Engel et al., 2008). A successful PES programme generating environmental services is not guaranteed to do so indefinitely. As payments are intended to be contingent on service production there is no reason to believe that, if the underlying externality is itself permanent, this service will continue to be provided once payments end (Wunder et al., 2008). Many farmers participating in the world's largest PES the Chinese Sloping Lands Conversion Programme, said they planned to reconvert land back to former uses once the programme was over (Groom and Palmer, 2012a). Given the basic premise of PES as a conditional incentive tool, there can be little expectation of permanence in the absence of payments. On the other hand, the very nature of voluntary participation in PES can assure permanence; it gives each party the ability to renegotiate in changing climates and markets (Engel et al., 2008; Pagiola and Platais, 2007). Given the limited operating time frame of most PES schemes, insufficient information is available whether indeed payments will be able to promote long-term land use changes, or indeed if payments must be continuous (Wunder et al., 2008).

2.3.2.5 Liability

Liability has been identified as a key precondition for permanence within PES schemes (Palmer, 2011; Sedjo and Marland, 2003). Put simply, liability can be described as having a high probability of being held accountable for – and penalised for – the failure to deliver those environmental services under PES contract (Palmer, 2011).

Who is liable if a PES does not deliver is an important question, and one that is still hotly debated. It has perhaps received the most attention within the carbon market and REDD+ (Angelsen, 2008; Dutschke and Angelsen, 2008; MacKenzie et al., 2012; Palmer, 2011; Phelps et al., 2011). One of the main issues in securing funds for the carbon market is this: under the Kyoto principles, liability for the loss of climate change benefits, e.g. reduced carbon emissions, is transferred from project developer to those purchasing the carbon credits (UNFCCC, 2005). In other words, once credits are sold, the sellers are no longer liable for any losses (or failure to provide environmental services), although they continue to maintain control over the resources via ownership and use rights (MacKenzie et al., 2012; Phelps et al., 2011)

The problem of liability is most acute at the project level, where cause and effect are harder to prove and/or prove difficult to measure (Kroeger and Casey, 2007; Palmer, 2011). These risks are higher where individuals, particularly the poor, cannot be directly sanctioned and when penalisation is merely the termination of future payments (Palmer, 2011), as is currently the norm. "Voluntary" agreements, such as these, require allocation of liability in such as way as to reduce the incentive to 'opt out' at a future date, and in particular before contract completion. Mechanisms such as co-management and nested liability frameworks have been suggested (Carlsson and Berkes, 2005; Palmer, 2011; Pedroni et al., 2009) which transfer a degree of liability to the resource owner. One concept of liability, which could transfer well to low-income user rights, is sanctions based on the removal of some of their use rights in addition to the termination of PES payments, which would be institutionalised under a co-management platform. In the event of non-permanence, enforced reductions in resource extraction ensures environmental service delivery above the reference level (Palmer, 2011).

2.3.2.6 Legitimacy and fairness

Legitimacy and fairness are now recognised as important attributes in the long-term success of PES, although more often than not, their consideration in project design requires a decrease in short-term efficiency (Corbera and Pascual, 2012; Muradian et al., 2010; Narloch et al., 2013). The legitimacy of any PES scheme will rely on stakeholder's preferences and perceptions of how the scheme conforms to local formal and informal rules and social norms (Gross-Camp et al., 2012).

Yet, for PES to be effective and efficient they need to contract with those who constitute a credible threat to environmental service provision, and to do so in a cost-effective manner (Wunder, 2005; Yang et al., 2013a). The question of who should get paid is, however, never this simple. In only paying those individuals generating

negative externalities, PES schemes can generate perverse incentives and moral hazard (Alpízar et al., 2013).

In essence, a PES project can be viewed as payments to polluters, the value of the payment depending upon the degree to which they pollute. As such, land stewards who already engage in effective environmental practices should not – from an efficiency point of view – be entitled to compensation (Salzman, 2005). However, exclusion of such players can be seen as unfair, and, evidence suggests, can even result in 'behavioural spillover'. Behavioural spillover means that those excluded from a new PES scheme choose to reduce their contributions to service supply even though they face no change in price or income, or even induce degrading practices in order to receive payments (Alpízar et al., 2013; Muradian et al., 2013; Wunder et al., 2008). This in turn can significantly influence long-term contributions to public good supplies (Alpízar et al., 2013).

From an efficiency point of view, PES should also be targeting larger resource users with economies of scale: generally those with larger holdings. However, as previously mentioned, excluding small holders can further exacerbate poverty and marginalisation and PES design should mitigate such effects (Grieg-Gran et al., 2005; Miranda et al., 2003). Schemes which enable PES benefits to be distributed across a wider range of stakeholders, including those non-participants, can go a way towards mitigating exclusion and perverse incentives (Mahanty et al., 2013).

Perhaps one of the most important determinants of legitimacy is the inclusion of key stakeholders in PES design, as has been suggested by many scholars (Corbera et al., 2007; Gross-Camp et al., 2012; Narloch et al., 2013; Sommerville et al., 2010a).

Views on equity and fairness are ultimately cultural and context specific and designing 'fair' and legitimate PES will prove challenging: there is no "one size fits all" (Narloch et al., 2013; Pascual et al., 2010). Context-specific views on equity must be addressed and taken seriously within PES design. If not, PES risk eroding intrinsic motivations and pre-existing institutions which can ultimately reduce the effectiveness of PES, if not undermine them completely (Muradian et al., 2013; Vatn, 2010).

2.3.2.7 PES suitability

PES schemes hailed as instruments which can solve environmental and development issues at the same time. Yet, there are numerous reasons why PES may not be the most appropriate tool. Under perfect market and institutional settings, a budget

constrained donor will always prefer PES to more indirect approaches, these being more cost-effective and efficient (Ferraro and Simpson, 2002). However, in reality constraints as well as market and institutional failures are the norm, particularly in low-income countries (De Janvry and Sadoulet, 2006; De Janvry et al., 1991). As such, the provision of credit, expertise or technology may in fact be the more cost-effective intervention (Groom and Palmer, 2012b). Severity of constraints, relative prices and the type of technologies have been shown to affect the cost-effectiveness of PES (Groom and Palmer, 2012b). How individuals respond to PES will be influenced by their ability to interact with other markets such as labour and resource markets (Muller and Albers, 2004). Combined policies which reduce constraints alongside PES, thus enabling PES to be more efficient, should not be overlooked in conservation's current market-centric approach (Groom et al., 2010). Subsidies can work alongside PES to reduce access barriers and constraints; but in addition can be a source of environmental degradation (Groom et al., 2010; Palmer, 2011). Elimination of such subsidies, i.e. those for inputs, should also be removed prior to investments into PES.

Where PES must contract with multiple sellers, transactions costs will inevitably rise. Contracting with groups or communities can theoretically reduce costs (Jack et al., 2008); this can include contracting with communities for the protection of common pool resources (CPR). CPR are those which are rival and non-excludable (or where exclusion is extremely expensive) (Fisher et al., 2010; Kemkes et al., 2010). While PES can learn much about contracting in these areas from previous CPR literature (Clements et al., 2010; Fisher et al., 2010; Muradian et al., 2010) – a description of which can be seen in Table 2.2 – contracting with groups will be inherently more difficult and subject to issues of leakage, free-riding and distributional constraints (Alix-Garcia et al., 2012; Sommerville et al., 2010a) as well as possibly capture by elites (Bennett, 2008; Sommerville et al., 2010a). In dealing with these issues – through design, screening and ensuring equity to name a few – transaction costs will inevitably increase, and may end up costing more than other less direct instruments.

Table 2.2 CPR management lessons for PES design and implementation

	CPR considerations	*PES analogs*
Resource size	The size of the resource and knowledge of its boundary are both characteristics that can enable better management. Although scale is relative, typically the smaller the resource the easier it is to carry out management principles.	Ecosystem services are delivered from nested processes. Microscale nutrient exchange affects macro-scale biomass production. As scale increases in a system so does the ecological complexity. This invokes the importance of subsidiarity.
Group characteristics	The size of the stakeholder group, the level to which they have shared norms, and interdependencies across groups are all characteristics that have been shown to affect CPR management. Smaller group size, more common histories and norms and more interdependent	Both the level to which ES buyers 'trust ' the providers to deliver a service, and the level to which the providers 'trust ' the PES scheme and its initiators have been shown to affect performance, implementation and legitimacy of PES. The number of participants (size) is also likely to affect cost of PES implementation and may therefore limit total ES delivery.
Resource–group relationship	When the resource is in close proximity to most of the stakeholder groups, better management is enabled. The level of dependence on the resource can affect its management — the higher the level of dependence the more incentive to manage it properly.	Where the resource providing the ecosystem service is far removed from the beneficiaries (e.g. carbon offsets), the verification of the effectiveness of intervention has been deemed critical for buyers.
Institutional arrangements	Governance rules must be clear in nature and seen as appropriate by the majority of stakeholders. Ideal rules are derived from inclusive processes and not be seen as 'top- down.'	Broad stakeholder inclusion, buyers and sellers of ES, should be involved in design and monitoring. PES contract allocation should be transparent, and the contracts themselves should have some flexibility to acknowledge changing opportunity costs.
Resource–institution relationship	Both an institutional knowledge of how the biophysical system works and spatial overlap between the processes that deliver the resource and the governing institution are enabling factors for successful management.	PES schemes should show that the intervention can deliver the service it promises. The scheme should be able to monitor system function over time, as well as monitor use, costs and benefits over time.
External environment	Exogenous factors such as demographic and technological changes can greatly affect the success of management. For example, improved low-cost technology for monitoring a resource may greatly enable management, where rapid population growth could hinder it.	Technological changes and population pressures (including changing preferences) affect how PES schemes are monitored, negotiated and implemented. Technology advances can make monitoring, compliance or defection easier; or for example, changes in global price of hardwood could affect local contract compliance.

Source: Adapted from Agarwal (2002) in Fisher et al. (2010)

2.3.3 A problem of demand

Such uncertainties in contract design and effective supply lead to a very different global picture of demand than that painted in Section 2.3.1. As yet too few buyers are

confident about the PES mechanism and its potential to deliver the stated services to be willing to commit to these emerging markets (Wunder, 2005).

Taking the example of the carbon markets, with a estimated total value of US$ 176 billion in 2011 (Kossoy and Guigon, 2012): it is only really the forestry subsector (under which umbrella mangrove forests can in theory be included) which has the potential to provide sustainable conservation finance alongside other environmental and social benefits. Yet, forestry is largely limited to the voluntary carbon market (VCM), where demand is driven largely by corporate social responsibility. In 2011, the value of the VCM was an estimated at US$ 424 million, substantially less than compliance carbon markets. While within the VCM the volumes of credits from the forestry sector have been increasing, REDD for example generated 29% of all VCM credits in 2011 (Peters-Stanley et al., 2011), the value of forestry credits is highly variable. REDD credit prices ranged between US$ 1 and US$ 25 in 2010 (Hamilton et al., 2010). The uncertain prices for forestry credits, uncertainty in liability and the overall small share of the total carbon market that this sector holds illustrate the limited finance for sustainable conservation that forest carbon markets provide (Grimsditch et al. 2012). In these times of climate change and growing environmental awareness, the fact that revenues for such carbon-related PES schemes are so limited can hold little promise for those other environmental services which achieve much less media attention.

Moreover, biodiversity is often harder to monetise than many other services: interconnections are complex and generally poorly understood. The services provided by biodiversity are numerous, generally intangible and rarely consumed by one clearly identifiable beneficiary (Landell-Mills and Porras, 2002). As such it is even more difficult to get local and, more specifically, global beneficiaries to pay directly. Despite significant advances in recent years, most payments from biodiversity services remain limited to NGO and government budgets and experimental at best (Turpie et al., 2008).

Within the marine environment the picture is even more confused. Although the marine environment provides numerous environmental goods and services, poorly understood flows and interactions as well as ill-defined property rights make marine PES highly complex and dynamic systems. Moreover, the generally common pool nature and large number of stakeholders will make contracting, monitoring and enforcement difficult and costly. As recently as 2008, no PES schemes for coastal or marine environments had thus far been implemented (Pagiola, 2008).

This said, private sector funding for biodiversity has grown significantly, much of which is currently funnelled through three of the largest global conservation organisations: the World Wide Fund; The Nature Conservancy; and Conservation International. With the private sector generally favouring *quid pro quo* approaches, conservation funding is experiencing a shift to more contingent, more accountable business-type approaches (Wunder, 2006), of which PES are a prime example.

Advances in PES understanding and design prove hopeful for marine PES. In the past, the lack of property rights has been cited as a barrier to marine PES. More recently, however, there has been a shift in gear around the discussion of property rights, mostly due to the growing interest in REDD+ and equity. Some scholars submit that instead of actual land ownership (or private property rights), participants need only have rights over the service flow or, indeed, *de-facto* use rights (Alston et al., 2013; Lyster, 2011). Community contracts can offer economies of scale which reduce transaction costs (Jack et al., 2008) and may also be a more effective method in dealing with the complex external relationships between neighbouring resource users, moreover identifying community structures and frameworks will also be key to ensure a fair, transparent and long-term programme. Moreover, previous lessons with CPR management do not preclude large marine areas from successful management but do highlight the critical need to better understand the resources and resource users (Fisher et al., 2010). Advances in technology will help reduce problems of costly monitoring (Alston et al., 2013; Game et al., 2009). With such dynamic systems, preventing leakage will require new and innovative responses. The development of appropriate controls, risk buffers and insurance mechanisms will reduce investor risk and concerns. Reducing these associated costs and risks of marine PES will undoubtedly improve the ability of marine PES to access a wider market.

In reality, much work is still needed to identify what demand does currently exist and how this will likely grow in the coming years, and in particular what efforts are needed to promote demand; such will be key to PES and indeed marine PES success.

2.4 PES as an instrument for poverty alleviation

Irrespective of issues in design and demand, the question remains: can rural communities, both inland and coastal, provide the desired environmental services? Under what conditions can they do so?

One of the most obvious supply constraints to a successful marine PES is an apparent lack of defined property rights over the seascape. In reality, *de facto* tenure does often

exist, be it through community access rights or governmental permit schemes. And the ability to access this apparent 'common property' is often dependent on informal conditions of socio-cultural origin, such as inheritance of rights of access, membership of a user group, gender, kinships, allegiances and ethnicity, as well as a fisher's original asset base (Béné, 2003; Geheb and Crean, 2003).

Successful marine PES will rely on adequate identification of all actors – winners and losers, as well as an accurate analysis of associated opportunity costs and compensation packages. Yet to date, too little is known about the supply-side dynamics, for example: what resource-use incentives are preferential; what institutional preconditions are required; as well as how these benefit transfers will affect local livelihoods in often remote, cash-poor communities (Wunder, 2007).

The degree to which lessons learnt within a terrestrial context can be transferred is unclear; in particular that information relating to tenure, benefits transfer and relative distributional implications. In the past, the assumption that rural coastal fishing communities are homogenous and function in a way similar to terrestrial counterparts has led to management practices which have generally proved unsuccessful. Perhaps to a greater degree, fishing communities are characterised by complex livelihood strategies and rich socio-economic and institutional networks than their terrestrial counterparts (Allison & Horemans 2006; Béné et al. 2000).

PES can offer new income sources in cash-poor areas and, when well administered, can provide a more stable cash flow than alternative sources such as cash crops. This should, in theory, enable a more flexible use of natural assets, diversification and greater livelihood security (Tschakert, 2007; Wunder, 2008). Indeed, a number of studies point to the success of PES projects as poverty alleviation tools.

In the Pimampiro watershed of Ecuador, Echavarría et al. (2003) found that participants received average payments of US$ 21.1 per month, representing an average increase of 15% in household disposable income; the larger Costa Rican PSA programme found that PES payments accounted for more than 10% of household income in over one quarter of participants (Ortiz Malavasi et al., 2003). Contributions can be even more profound in poverty struck areas. Again, in Pimampiro Ecuador payments for watershed protection to poor upland settlers comprised 30% of household spending on food, medicine and schooling (Echavarría et al., 2003). In Costa Rica's Osa Peninsula, a small survey found that 50% of participants below the poverty line were lifted above it due to the scheme and PES revenue became a primary source of cash income for 44% of these households (Grieg-Gran et al., 2005; Wunder, 2008). However, these studies provide only gross figures for PES income

and ignore landowner opportunity costs. Under such conditions it is anticipated that the gains would be much more modest. Indeed, more recently a review of seven PES schemes by Mahanty et al. (2013) indicated that although schemes did show small additional income to participating households, payments were often insufficient to cover the opportunity costs to participants over the lifecycle of the programme.

Recently, work on the Chinese Sloping Lands Conversion Programme (SLCP), the largest PES scheme in the developing world, has raised some interesting insights into PES as a poverty tool. The SLCP was first implemented in 1999 in response to human, agricultural and fishery losses associated in part with the deforestation of upland river basins. As a set-aside cropland programme, its main aim was to prevent soil-erosion though the afforestation and/or reforestation of highly sloped cultivated lands in the upper areas of the Yellow and Yangtze river basins (Groom and Palmer, 2012a; Uchida et al., 2007). By 2005 over 9 million hectares of cropland had been retired by over 15 million farmers and included lands in 25 provinces and municipalities in China (Bennett, 2008; Groom and Palmer, 2012a; Uchida et al., 2007); environmentally the SLCP has been considered a success (Groom and Palmer, 2012a).

A secondary objective of the SLCP was poverty reduction, the long-term successes of which remain less obvious. In the first instance, Uchida et al. (2007) found that participating households had lower initial levels of income, lower house values and asset holdings more generally than non-participating households. Using three methodological approaches – propensity score matching, differences-in-differences and differences-in-differences matching – the authors present evidence for a positive welfare effects from the programme on rural households; income from livestock and certain asset holdings increased significantly more so for participating households than for non-participating households. However, Uchida et al. also warn that the programme does not systematically favour the poor and only minimal statistical differences between the poor and better-off participating households are seen, hence there are few positive distributional impacts. In fact, Uchida and co-author's research hints that, in the future, richer households will be better placed to achieve structural change, promoting long-term income streams from on-farm to off-farm income sooner due to less current constraints and a less urgent need for immediate profits. However, more recently, Groom (2012, In: Groom and Palmer, 2012b) and Li et al. (2011) show significant income effects for participants from the SLCPs and that this result is in fact more profound at the lower quartiles of income distribution.

Many scholars warn against assuming the pro-poor advertisements of PES (Corbera and Brown, 2010; Grieg-Gran et al., 2005; Kerr, 2002; Kosoy and Corbera, 2010; Landell-Mills, 2002; McAfee and Shapiro, 2010; Pagiola et al., 2005).

In the first instance, PES schemes can only assist in poverty reduction in those areas where service production exists, is at risk and overlaps with incidences of high poverty (Tschakert, 2007). However, where high environmental benefit, low opportunity costs and high poverty overlap PES programmes can theoretically achieve dual goals cost effectively (Gauvin et al., 2010). Yet, while impoverishment and vulnerability are have long been recognised within coastal communities, poverty reduction is by no means guaranteed.

Many assume that PES schemes will contribute to the alleviation of poverty through the payments made to poor land/resource owners; indeed the premise that participation is voluntary creates a presumption that potential actors will simply refuse to participate or withdraw if benefits are not realised (Pagiola et al., 2005). However, in reality, PES may not be voluntary at the individual level (Bennett, 2008; Sommerville et al., 2009; Uchida et al., 2007). PES programmes, in an aim to obtain efficient outcomes through community contracts, can reinforce existing power structures and inequalities, particularly with respect to resource access rights (Pascual et al., 2010; Sommerville et al., 2010a). At a global level, there are concerns that REDD+ can reverse recent trends of decentralised natural resource management, placing power – and carbon rights – back into the hands of more disconnected governments (Phelps et al., 2010). Moreover, participation can be blocked for many poorer households by significant up-front financial and labour costs (Mahanty et al., 2013).

More generally, no results pertain to the wider community of poor. In fact, the PES literature remains virtually silent on issues of distribution within these schemes (Landell-Mills and Porras, 2002; Pascual et al., 2010). Of notable exception is work by Sommerville et al. (2010a) who provide evidence of inequitable benefit distribution and elite capture mentioned above. However despite growing interest, empirical evidence on the extent to which PES schemes increase household welfare amongst participants and particularly those implications for non-participants remains scant; to date, anticipated benefits for the rural poor remain more hypothetical than real (Engel et al. 2008; Pagiola et al. 2008, 2005; Tschakert 2007; Wendland et al. 2010; Wunder 2006).

More generally, Wunder et al. (2008) indicate that in the majority of cases, PES schemes have had some positive but point-wise, quantitatively small poverty

reduction effect, although gains are seldom huge. This said, the author also states that it is unlikely that PES schemes, given their current small scale, contribute to huge poverty exacerbation effects, nor does he believe PES will contribute significantly to poverty alleviation (Wunder, 2005).

In practice, PES schemes have the potential to affect three categories of 'poor'. These categories include (a) on the supply side: poor environmental services sellers (project participants), (b) on the demand side: poor environmental service buyers and (c) derived effects: other poor potentially impacted by the implementation of the PES.

On the supply side, the question is what participation filters exist which may prevent participation of the poor?

2.4.1 To what extent are the poor able to participate in PES?

The poor face explicit PES access rules and underlying structural impediments. A number of barriers constrain the successful participation of actors within PES schemes and in general it is those 'poorer' members which are largely left out. Wunder (2008) identified four selection criteria constraining the participation of the poor within PES. These include: eligibility; desire; ability and competitiveness.

Many of the facets preventing eligible enrolment, such as insecure tenure, lack of access to credit, lack of information, lack of title and small land holdings are often directly correlated with poverty (Grieg-Gran et al., 2005; Mahanty et al., 2013). Moreover, poorer and more vulnerable families often have fewer diversification options, constrained by a smaller asset base (Grieg-Gran et al., 2005; Landell-Mills and Porras, 2002; Pagiola et al., 2005; Tschakert, 2007; Wunder, 2008).

If eligible, the poor also require the ability to enrol. For households that rely entirely on subsistence living, setting aside land or resources may not be feasible. Landholders owning plots of only a couple of hectares, or those fishing for subsistence only, would find it almost impossible to set aside resources predominately for environmental service production. Studies from Ecuador and Guatemala found farmers with the smallest landholdings less willing to participate in a PES set-aside scheme, believing it to compromise food security (Grieg-Gran et al., 2005; Southgate et al., 2009). In those instances where active investment is required, conversion to more biodiversity-friendly agriculture practices of fishing practices, poorer households may lack the necessary skills, labour or capital (Southgate et al., 2009; Wunder, 2008). When these new practices are complex, accessing technical assistance can prove harder for those poorer households (Pagiola et al., 2008).

Taking eligibility, desire and ability together it can be seen that there is a serious risk that PES schemes can in fact exacerbate impoverishment, increasing the relative poverty of those unable or unwilling to participate. Indeed, the overall conclusion of an extensive literature review conducted by Centre for International Forestry Research (CIFOR) concluded that concerns should sit with the non-participating poor (Wunder, 2008).

Recent evidence suggests that to a large extent the poor are often not able to fit the criteria. Miranda et al. (2003) show that in the Virilla PES watershed project in Costa Rica, three quarters of participants already earned over US$ 820 per month, thus were hardly the poorer members of society in the first place. Information on the 110 landowners receiving payments indicated that over 80% owned properties of over 70 hectares; a mere 6% of enrolees owned properties of 30 hectares or less. In fact, most landowners taking part were not dependent upon their land, and 65% were either professionals, employed in trade or retired (Grieg-Gran et al., 2005; Miranda et al., 2003). A larger study by Zbinden & Lee (2005) found a similar pattern within the northern lowlands of Costa Rica. On average, participants were found to be better educated, typically urban dwellers and proportionally reliant upon off-farm sources. Participants were also more likely to own larger farms than nonparticipants (Zbinden and Lee, 2005). More recently, analysis of an agro-forestry PES scheme located in the Sofala Province, Mozambique found male-headed and higher income households were favoured as project beneficiaries (Hegde and Bull, 2011).

However, Uchida et al. (2007) found that participating households in the Chinese SLCP had, in general, lower initial levels of income, house values and asset holdings than non-participating households. Within a Nicaraguan silvopasture PES project, Pagiola et al. (2008) found poorer households able to participate extensively within the aforementioned scheme, in some cases to a greater extent than their wealthier counterparts. Moreover, participation was not limited to the simpler, least expensive interventions. Poorer households accounted for a 51% and 70% decline in degraded pastures and area under annual crops respectively; a substantial share of land use changes. The extreme poor appeared to display a slightly greater degree of difficulty in participating, although this difference is relative (Pagiola et al., 2008).

2.4.2 To what extent do the poor realise benefits?

Once eligible, willing and able to participate, the question is whether PES actually do make those poorer service providers better off? And if so to what extent do they actually benefit?

The commodification of environmental services is often structurally skewed against the interest of local stakeholders and tends to favour elites (Corbera and Brown, 2010; Kosoy and Corbera, 2010; Mahanty et al., 2013; McAfee and Shapiro, 2010). Poor landowners generally own marginal lands of low productivity and lack a strong voice to best negotiate contract rules and payments (both at the local and contract level) (Wunder, 2008).

The realised benefits of any PES will of course vary, and are perhaps dependent upon the degree to which poverty alleviation is considered within the remit of project objectives. Evidence from past poverty reduction interventions suggest that geographical targeting and self-selection criterion found within PES schemes can contribute to their pro-poor performance. A review of targeted poverty reduction programmes found projects that used geographical targeting and self-selection achieved higher rates of transfer to lower income households (Coady et al., 2004).

Recent evidence does indeed suggest that poor households can benefit financially and otherwise subject to favourable contract design and institutional conditions (Mahanty et al., 2013). However, worryingly, realised benefits often prove to be dependent on ownership of a sufficient asset base in the first instance (Clements et al., 2010; Jindal et al., 2010; Mahanty et al., 2013; Sommerville et al., 2010a; Wunder, 2008).

Where PES are appropriately designed to enable adequate access of the poor, its success to lift participants out of poverty will ultimately rely on its ability to promote long-term diversification into alternative occupations and behavioural change (Uchida et al., 2007; Xu et al., 2004). PES have been theorised to further influence poverty through promoting alternatives e.g. 'off-farm' labour (Groom and Palmer, 2012a, 2012b; Groom et al., 2010; Kelly and Huo, 2013; Uchida et al., 2009). Some suggest that this is simply through a simple labour substitution mechanism whereby PES merely reduces the need for on-farm work (Kelly and Huo, 2013). Others suggest more complex relationships. Recent analyses suggest that for credit constrained households PES schemes can promote diversification and poverty alleviation through relaxing liquidity constraints, for example when a household does not have the means to finance its shift into another more profitable market. Research into the SLCP suggests that through the relaxation of credit constraints, many participants have indeed reallocated household members' time to more lucrative off-farm work, and that they did so at a higher rate than non-participants (Uchida et al., 2009). High transaction costs, insecure tenure and other constraints have also been shown to inhibit off-farm labour, self-employment and the seeking of wage-earning jobs (Groom and Palmer, 2012a; Uchida et al., 2009). Evidence also suggests that

constrained and unconstrained households will react differently to PES schemes and potential impacts will be different; for those unconstrained households poverty reduction will only occur as a result of temporary compensation since their allocation to off-farm labour did not change (Groom and Palmer, 2012a; Groom et al., 2010). Further evidence provided by Groom et al. (2010) also indicates that constrained and unconstrained households exhibit differing responses to the SLCP intervention. Moreover the various constraints themselves, such as access to credit, markets, technology, household composition, tenure and land quality to name a few will have an impact (Groom and Palmer, 2012a; Groom et al., 2010; Liang et al., 2012); a better understanding will hopefully enable a better targeted, more cost-effective policy. Indeed, as well as improving overall efficiency, working alongside other instruments and policies can assist in relaxing those constraints which at present reduce the ability of such households to participate (Groom et al., 2010).

In addition to fiscal benefits, which may in fact not be substantial – PES schemes can have further more profound benefits. PES programmes have been shown to provide more stable incomes as well as numerous non-income benefits. Participation within a PES scheme has, in some cases, the potential to increase smallholder tenure security, particularly against neighbours and squatters. Local acceptance of tenure is often dependent upon economic use of land or resource. By creating local recognition of the tangible income-generating value of conservation areas and practices, resources can become less susceptible to grabbing, as was witnessed in the Bolivian Los Negros PES pilot programme. Participants received maps with demarcated boundaries helping demonstrate the economic value of the 'idle' land, giving higher *de facto* protection from landless migrants (Pagiola et al., 2005; Wunder, 2008).

Beyond tenure consolidation, PES schemes have been shown to increase human and social capital. PES can strengthen local community institutions as well as support the development of new ones. Payments can fund the cost of management by village institutions, particularly over common-pool resources, including such things as monitoring and sanctioning non-compliance (Clements et al., 2010). Furthermore, PES schemes which require technical inputs bring training and new skill sets, often up-front, allowing diversification.

2.4.3 Implications of project design for poor within PES

Variations in PES structure and design include: environmental services provided; location; eligibility rules for participation; payment or incentive type; as well as the composition and social norms of different stakeholders. The possible effects for all involved will very much depend upon all of these factors. Projects may not reduce

local poverty if they in fact simultaneously reduced availability of a staple crop, thereby increasing its price beyond that of basic income gains.

It goes without saying that if PES programmes wish to include poverty alleviation in their manifesto, effort is needed to understand and reduce those initial barriers which inhibit and/or constrain the participation of the poor; lack of capital remains a significant barrier to access. Moreover, PES schemes continue to reduce informal access to many resources and understanding possible land-use and ownership reform is imperative in order not to exacerbate existing inequalities (Mahanty et al., 2013).

Beyond this, however, the type of project implemented can exacerbate or limit equity implications of PES. Although ex-post studies do not appear to exist, modelling by Zilberman et al. (2008) indicates that two different PES scheme types can have very different outcomes; PES which diverted land vs. working-land programmes resulted in different implications for a range of stakeholders. Analysis assumed that the PES would affect two outcomes – an agricultural good and an environmental service - and affect four groups: rural landowners, rural landless, urban consumers and beneficiaries of the environmental services. Working-land schemes were mostly found to have better distributional effects than land-diversion programmes. This was mainly due to the generated employment opportunities; setting areas aside generally having the opposite effect. Zilberman et al. also found that the landless were most likely to gain from PES schemes which led to both higher output prices and wage rates but where agricultural product had a relatively small share in their overall consumption set. Under targeted payments, small unit landowners with low land productivity and limited potential for supplying environmental services will gain little rent and likely be affected similarly as the landless poor.

Again, the overall implications of all schemes depended on effects on food and living expenses relative to benefits received, as well as potential multiplier effects (Zilberman et al., 2008). Currently, dramatic effects appear quite unlikely, as areas enrolled within PES schemes are relatively small (Pagiola et al., 2005).

2.5 Merging PES into a marine and coastal context

Policy development for fisheries management has proved problematic. Sustainable fisheries continue to be the exception rather than the rule. Growing populations and unregulated coastal zones mean that most coastal artisanal fisheries are now considered overfished or collapsed (Defeo and Castilla, 2005; Hawkins and Roberts, 2004).

MPAs are the most prolific management tool in virtually all the world's oceans and seas (Agardy et al., 2003) and are touted as the most efficient management tool for overexploited fisheries within low-income tropical countries (Pollnac et al., 2001).

2.5.1 Marine Protected Areas

MPAs take many forms, for example no-take zones, multiple use, temporal closures and can be implemented at a national, regional or community level. However, they have the common characteristic of being a management intervention that is spatially organised (Christie and White, 2007).

Clear evidence exists that MPAs can improve fisheries health; in the last five years new, rigorous and defensible evidence has emerged which shows that MPAs do improve fishery yields and conserve biodiversity (Agardy, 2000). Today, virtually all coastal countries have implemented some form of MPA (Agardy et al. 2003) and in 2002 and 2003 respectively the World Summit on Sustainable Development and World Parks Congress called for the establishment of a global system of MPAs (Balmford et al. 2004, Kelleher 1996).

Despite this, the majority of MPAs worldwide show disappointing levels of compliance (Beger et al., 2005; Depondt and Green, 2006; Hargreaves-Allen et al., 2011). An initial broad assessment in the mid 1990s found that only one third of 383 MPAs had met their management objectives, a further third only partially and the remaining third not at all (Hargreaves-Allen et al., 2011; Kelleher et al., 1995). More recently, of 400 MPAs in the Philippines only some 20-25% are considered successful; furthermore, over 66% and 90% of Caribbean and East Asian MPAs respectively have failed to reach their management goals (McClanahan 1999; Pollnac et al. 2001).

Poor design, lack of appropriate scientific evidence and disregard for social contexts of many vulnerable stakeholders has led to the failure of many MPAs to meet their management objectives. Conflicts arising from the economic dislocation and marginalisation of artisanal fishermen are not uncommon, and lead to the rejection of the imposed MPA or loss of interest after initial support (Christie, 2004).

MPA success is, in the large part, dependent upon local community and stakeholder involvement; research highlights social factors, and not biological or physical variables, to be the primary determinants of MPA success or, indeed, failure (Mascia 2003). In particular, fisher co-operation and their recognition of MPA boundaries are integral. In most cases, rebuilding stocks requires a significant reduction in fishing effort, at least in the short term. For those with limited means to buffer these losses, costs can be particularly high (Mohammed, 2012). Alternative occupations have been

recommended for fishers displaced due to catch scarcity or displaced by MPAs (Sievanen et al., 2005; Smith et al., 2006; Teh et al., 2008; Worm et al., 2009). Indeed, Pollnac et al. (2001) found the presence of successful alternative income projects among one of the six most important indicators of successful community-based MPA management.

Under the current situation it is highly unlikely that MPA budgets will stretch to cover the opportunity costs of displaced fishermen. In truth, insufficient funds for effective basic MPA management costs are the norm worldwide (Depondt and Green, 2006).

However, the limited success of MPAs has not led to a reduction in their implementation; MPAs continue to become the mainstream management tool in virtually all the world's oceans and seas (Agardy et al., 2003). Other interventions used such as catch limits remain unsuitable and costly in an artisanal developing world context due to dispersed landing sites and multi-species catch.

PES can provide additional funds outside of government budgets, donor funding and trust funds: traditional MPA funding sources. Under such an approach, beneficiaries of MPAs can deliver the funds required to compensate fishers for forgone opportunity costs. Alternatively PES can circumvent the need for MPA establishment providing direct compensation for resource protection.

2.5.2 Implications for poverty in a marine and coastal setting

Although advocated as having the ability to address both conservation and poverty concurrently, as previously described, few PES schemes have been carefully documented. Very little is known about the possible distributional implications within terrestrial PES schemes, much less within a marine and coastal setting where PES programmes remain in the proposal stage (Engel et al., 2008; Pagiola et al., 2008).

In the first instance, can PES contribute to community development and poverty alleviation? To what degree will fishers wish to participate? To what degree will poor fishers wish to participate? What factors may prevent their participation? Given the troublingly low compliance rates within MPA design, will the sole inclusion of an incentive package inspire compliance and behaviour change? How best can schemes be designed in order to promote participation?

As previously mentioned, some claim that PES will in fact increase relative poverty in places; under a terrestrial PES scheme, landless poor may become relatively worse off as profits may be restricted to only those with tenure rights over resources. Further

barriers to PES market access may exist based on gender, kinship or other divisions (Landell-Mills and Porras, 2002; Wunder, 2005). It is likely that many lessons can be learnt from terrestrial PES schemes but significant differences do complicate matters within coastal communities. Fishermen have long been thought of as the 'poorest of the poor', and fishing as a last resort. However, fishing communities are far from homogenous units; important differences exist between individual resource users regarding access, assets, knowledge and institutional linkages (Tschakert, 2007); all resource users are not equal. Indeed poverty, and relative degrees thereof, are multifaceted and vary widely within fishing communities; further discriminations, marginalisations and exclusions exist within fishing communities (Jentoft et al., 2010).

Béné (2003) identifies four categories of discrimination within fisheries systems: economic exclusion, social marginalisation, class exploitation and political disempowerment. These mechanisms are described in Table 2.3. The ability of fishers and coastal peoples to adopt marine PES programmes will be further influenced by one, few or all of these discriminations. It is important that one can identify those more vulnerable fishers whom may not, for some reason or another, be eligible or desire to participate.

In the second instance, can demand successfully fund such schemes? What are the issues surrounding the realisation of this demand? As yet, few buyers are confident about PES and its potential to deliver the environmental services promised (Wunder, 2005). This problem may be more profound in a marine setting where weak tenure exists and environmental services are more diffuse, fragmented and to a large extent 'invisible' (Pagiola, 2008). However, recent interest has been shown by such organisations as Forest Trends[3], a number of NGOs including The Nature Conservancy and CARE, as well as the Mexican Government[4] (Muñoz 2009, pers comm.). Furthermore, blue carbon is emerging as an important market for marine PES (Murray et al. 2011).

[3] Forest Trends is an interdisciplinary not-for-profit organisation initially focussing upon market-based approaches for forest conservation. The organisation now also looks at such practices for marine conservation through its marine branch MARES.

[4] The Mexican Government is in the early stages of implementing a programme which attempts to reduce the fishing mortality of the endangered vaquita dolphin. Payments are made to fishermen to incentivise the uptake of gear more 'vaquita friendly', payments are intended to cover the opportunity costs of associated losses due to the use of new nets which, while enable vaquita to more readily escape from the nets, are less effective at catching fish vs. the previous gillnets. Entry into the fishery is regulated by a licence cap.

Table 2.3 Typology of intrasectoral "socio-institutional" processes leading to the denial of individuals or groups' commands over the resources

Discrimination process	*Definition*	*Access denied*	*Nature of discrimination*	*Comments*
Economic exclusion	Process which leads to the leaving out from a particular economic activity of certain individual due to their economic/financial inability to access the factor of production necessary to enter and/or operate this activity	Yes	Economic	
Social marginalisation	Process which leads to the denial of the command over a resource, service of commodities for certain actors based on such criteria such as caste, gender, or ethnic origins	Yes	Social	Like exclusion, marginalisation refers to situations where actors are denied access and use of a resource (as opposed to class exploitation). The two concepts, however, differ by the nature of the barriers: economic in the former social in the second
Class exploitation	Situation where a higher class is perceived as being in the position of extracting surplus labour from a (lower) working class is considered as not receiving its "fair share": in the benefits created by an economic activity	No	Social	Exploitation differs from exclusion and marginalisation in that it corresponds to cases where the poor are not denied access to the resource/economic activity
Political disempowerment	Situation where actors are left out: from participation and/or decision-making processes leading to low/poor opportunities to control and govern their own commands over resources. This may result in reduction or even denying of access and use of the resources. The initial barriers are due to asymmetrical power relationships based on social stratification	Yes/No	Political (power)	Under situation of disempowerment, actors (users) may access the resources (e.g. the fisheries). What they do not access is the decision-making process (e.g. management system) which govern the modalities of access to and use of these resources

Source: (Béné, 2003).

2.6 Summary of key findings and research questions

However hampered by teething problems, the idea behind PES remains a solid one; adequate compensation mechanisms for conservation interventions are long overdue. A more equitable approach has lead to an uptake of PES projects within the conservation portfolio, particularly those projects which aim to address such supplementary goals as poverty reduction. Up until recently, as can be seen by available data, these projects have remained limited to terrestrial schemes providing watershed and carbon services. However, within the last year, there has been a growing interest in the use of PES within a marine context.

Given the preceding discussion, the question remains: what is the feasibility of marine PES? To what extent can we address the issues raised within a marine context?

With a greater focus on social responsibility within conservation, PES instruments are likely here to stay. Yet as can be seen, there is much to learn surrounding marine PES and their feasibility as a conservation tool, and the rapid uptake of PES schemes by development practitioners in recent years requires urgent investigation.

In reality, a PES scheme may often not be the optimal mechanism. Marine and coastal environments are very different from their terrestrial counterparts. Understanding the similarities and these differences, as well as the possible distributional implications of marine PES schemes will be integral to the future success of marine PES in meeting their goals.

Identifying those instances where marine PES will be applicable, and indeed where they are not, where they can contribute to rural poverty and where they can work alongside pre-existing tools will be important to their realisation within the marine setting. Furthermore, it will be central to their long-term success; if marine PES do not quickly adapt to the marine setting appropriately it is likely that they will be rejected by communities before their foot is even in the door.

Chapter 3

Examining the issues in marine Payments for Environmental Services through expert elicitation

3.1 Overview

In contrast to the previous research papers, here we focus on issues associated with particular promoting confidence in marine PES instruments more generally. The paper uses expert interviews collected via web surveys to highlight benefits and perceived challenges in developing the tool for the marine environment. As expected within a marine context, ill-defined property rights features as a prominent barrier. Other obstacles relate to the more fluid and invisible nature of the marine environment and issues of monitoring and enforcement. However, we challenge these perceived limitations and discuss opportunities for marine PES to add to the current marine conservation portfolio. The results are widely applicable due to continuing high levels of dependence on and degradation of marine resources globally, as well as the recent growing interest in marine PES schemes.

The following section presents an introduction the paper. Section 3.3 introduces the current status of marine PES as well as the use of expert opinions in qualitative research and decision making for policy. Section 3.4 explains the methods used and Section 3.5 the expert profiles. Section 3.6 presents questionnaire findings. We conclude with the discussion and conclusion in Section 3.7 and 3.8 respectively.

3.2 Introduction

Over the last few decades, increasing pressure and a high dependence on coastal and marine ecosystems has seen the development of policy and legislative instruments which seek to protect, conserve and manage these resources (Borja et al. 2008). In the past, many countries have attempted to implement regulatory instruments to promote more sustainable use of marine resources, for example through no-take zones, restriction of fishing gears or via implementation of fishing permits (Mohammed 2012). However, these instruments have largely failed to extinguish unsustainable practises, particularly among those coastal communities located within low-income countries. For the most part, regulations do not adequately compensate for loss of earnings or inspire compliance, and enforcement proves difficult across multiple landing sites (Mohammed 2012).

Recent years have seen the emergence of a new tool in the conservation portfolio called 'Payments for Environmental Services' (PES). The premise of PES emerged as a solution to realign private and social costs resulting from land and resource use changes (Jack et al. 2008). The PES framework states that natural resource users are paid to conserve or manage natural resources more sustainably. The notion is that PES represent a win-win situation, able to improve compliance through compensation. As such, PES appear to be continually acknowledged as an alternative to failed regulatory mechanisms (Mohammed 2012), and more recently have seen a growing interest from the marine conservation and development arena. This has lead to a rapid adoption of PES, unfortunately with little critical discussion or analysis of long-term impacts (Redford & Adams 2009). However, as with any conservation instrument, implementation is far from simple.

Much uncertainty still surrounds PES, and to a greater degree marine PES schemes. Bringing these instruments to the coastal and marine environment represents a complex challenge. However, to date there has been little critical analysis of marine PES, both empirically and hypothetically. A lack of tangible examples limits the literature on marine PES to discussion pieces and policy briefs.

The purpose of this paper is to better understand the benefits and challenges of bringing PES to the coastal and marine environment. The paper presents the findings of questionnaires conducted with 42 experts in the field of PES and/or marine conservation and management. Results are based upon present understanding of the current marine environment and PES experience. In this final chapter a qualitative methodology is utilised. A qualitative approach enables useful insights to be drawn out from the findings, highlights diversity and, importantly, allows experts to frame their thoughts unprompted. The qualitative results presented herein also compliments previous quantitative chapters, drawing on a different but equally informative methodology.

The results are widely applicable due to a global dependence on marine resources, their ever-growing degradation and the recent flourishing of PES schemes within this marine context.

3.3 Marine PES and expert elicitation for policy design

3.3.1 Current status and knowledge on marine PES

The environmental services produced by coastal and marine systems have been well documented; these are summarised in Table 3.1. However, while PES programmes

have established themselves within the terrestrial conservation portfolio, marine PES instruments remain very much in their infancy.

Table 3.1 Services provided by coastal and marine ecosystems

	Coastal Ocean	*Open Ocean*
Provisioning Services	Fisheries & aquaculture	Fisheries & aquaculture
	Fuel wood	Alternative energy
	Alternative energy	Strategic & other minerals
	Natural products	Natural products
	Genetic and pharmaceutical	Genetic and pharmaceutical
	Transportation	Transportation
Regulating Services	Weather regulation	Weather regulation
	Carbon sequestration	Carbon sequestration
	Shoreline stabilisation	Nutrient regulation
	Natural hazard protection	Waste disposal
	Nutrient regulation	
	Waste disposal	
Supporting Services	Soil formation	Nutrient cycling
	Photosynthesis	Primary production
	Nutrient cycling	
Cultural Services	Tourism	Tourism
	Recreation	Recreation
	Spiritual values	Spiritual values
	Education	Education
	Aesthetics	Aesthetics

Source: Forest Trends & The Katoomba Group (2010)

More recently, discussion of such markets within the marine context has emerged in the grey literature (e.g. Forest Trends & The Katoomba Group 2010; Mohammed 2012; Pagiola 2008). Marine PES programmes remain for the most part in the proposal stage. With the rise of global climate change concerns, the inclusion of mangroves within the carbon mitigation mechanism REDD+ has seen a recent flurry of attention, including media and policy (Murray et al. 2011; Weaver 2011; Zwick & Kett 2010). Beyond 'blue carbon', marine PES schemes have also been proposed to improve coastal water quality as well as to promote compliance and compensate for restrictions to fishing areas (Begossi et al. 2011; Lindahl & Kollberg 2009).

Lindahl & Kollberg (2009) discuss the potential of mussel farming to reduce ocean acidification. Under an extension of the EU agri-environmental aid programme the authors argue that mussel farmers could be paid support for operations which as a consequence reduce eutrophication from nearby sewage plants. In Brazil, a payment scheme operates within artisanal fisheries. Called the defeso, fishers receive a 'salary' based on the minimum wage which compensates for opportunity costs lost during a

closure period whereby fishing is prohibited by the government for fish reproduction (Begossi et al. 2011).

A recent International Institute for Environment and Development (IIED) briefing paper by Mohammed (2012) discusses the application of marine PES as a mechanism to compensate fishers for loss of earnings; to induce restoration of coastal habitats; to incentivise protection of endangered species and to promote sustainable fishing practices. In their 2010 report *'Payments for ecosystem services: Getting started in marine and coastal ecosystems: a primer'* Forest Trends & The Katoomba Group highlight the possible role of PES in the marine environment. More specifically they mention marine carbon sequestration and capture, water quality and pollution filtration, shoreline protection and stabilisation, marine biodiversity protection and fish nursery habitat protection.

Within the academic literature, terrestrial PES are by no means presented as a panacea for conservation. Authors speak of complexities in design, implementation and equity (e.g. Corbera et al. 2007; Engel et al. 2008; Ferraro 2008; Gibbons et al. 2011; Jack et al. 2008; van Noordwijk et al. 2007) as well as enabling environments (e.g. Benítez et al. 2006; Engel & Palmer 2008; Engel et al. 2008). In addition, ex-post analyses of environmental success, economic efficiency and/or distributional implications remain scarce (Pascual et al. 2010; Wunder 2008). More generally, the application of PES schemes has run ahead of a sound understanding of the appropriate tools for effective implementation; few design recommendations are, or indeed can be, drawn from empirical evidence (Milne & Adams 2012).

For the greater part, the literature pertaining to marine PES schemes is in a practical sense non-existent. And while marine PES can learn some lessons from already limited literature of their terrestrial counterparts, marine systems exhibit fundamental differences which may have further ramifications for their success. Marine ecosystems have several characteristics unlike those found on land. Marine systems are highly mobile making them potentially harder and more expensive to monitor (Mohammed 2012; Pagiola 2008). Another, more pertinent, issue also stems from this trait: these dynamic systems can make causation difficult to prove. Unlike in watershed PES, where benefits roll downstream, in marine environments benefits are multi-directional and widespread. As such, identifying demand may prove difficult (Begossi et al. 2011; Pagiola 2008). While not exclusive to the marine environment, marine systems are also subject to ill-defined and insecure tenure, as well as multiple and fragmented resource users, which can make identification of appropriate suppliers more challenging.

3.3.2 Expert opinion in policy design & decision-making

Increasingly, expert opinion has been sought to inform decision-making within new policy design, particularly in cases of high uncertainty or where data is lacking (Lowe & Lorenzoni 2007; Weible 2008).

Expert judgement is not intended as a substitute for empirical research. However, it can provide useful insights for researchers, practitioners and policy makers, as well as raise awareness of potential issues while research for definitive results is on-going (Granger Morgan et al. 2001; O'Neill et al. 2008), highlighting benefits and challenges in new policy design. Indeed, specialised knowledge and expertise are considered important factors in any decision-making process, have been used extensively to solve problems related to environmental hazards (González et al. 2007; Morgan & Henrion 1992), and feature regularly within the political sciences (Dorussen et al. 2005).

Expert interviews can be a key strength in aiding design processes and highlighting future research needs. Results can be qualitative or quantitative (Hagerman et al. 2010). Expert elicitation does not need to identify consensus, but instead can highlight and voice diversity in thoughts and opinions, and make new knowledge available. Furthermore it is available almost immediately and unlikely to be reduced on the time-scale relevant for policy formation (González et al. 2007; Hagerman et al. 2010; O'Neill et al. 2008).

The complexity of many environmental systems and an urgent need to address issues of degradation mean that, often, expert knowledge and experience is the best evidence available (Fazey et al. 2006). Indeed, qualitative methods are being increasingly used in global assessments of marine conservation instruments (Balmford et al. 2004; Hargreaves-Allen et al. 2011; Hockings 2003).

As we are interested in the various views and technical issues of bringing PES to the marine environment, we consider expert elicitation an appropriate methodology for this study.

3.4 Methods

3.4.1 Expert selection

Marine environments are complex systems, as is the practical implementation of PES schemes. In order to fully understand the issues surrounding the transfer of PES to the marine environment, expert elicitation was not limited to solely PES experts. Those working in marine conservation hold important insights to the applicability of

PES in a marine context, and to a greater degree their experiences are not within the literature. For this reason we purposely sought the opinions of individuals working in one or both the field of PES and marine conservation, and included academics and NGO practitioners alike.

Expert opinions are elicited on the suitability of PES as a policy tool within the marine environment. There is no agreed definition underpinning what constitutes an 'expert' in the expert knowledge elicitation literature (Lowe & Lorenzoni 2007). In our case, we consider 'experts' to be those individuals with specialised knowledge on the topics PES and/or marine conservation management. Expert criteria included demonstrated experience in PES or marine conservation research as indicated through academic papers and/or PES or marine policy development and/or PES or marine policy implementation.

Expert selection was carefully considered, as this inevitably affects survey outputs. Individuals were identified through a review of the literature (academic and grey), review of relevant NGO programmes and key actors in design and/or implementation, referrals from experts themselves and the author's own personal knowledge of the fields. Experts were also solicited to take part in the survey via notification on the Coral-List forum[5]

Participating experts included leaders in the field of PES and marine management, as well as experts with specialised and/or practical expertise within one or both of these fields. In some cases expertise overlapped. A small number of respondents indicated no publications or field experience within the aforementioned disciplines. Those individuals with no publications or field experience and which were not individually solicited (e.g. from the web-search) were omitted from analyses. However, a number of these respondents were retained as they were still considered as experts due to publications and/or vast practical experience in a closely related field and carried valuable contributions.

3.4.2 Questionnaire design

Expert opinions were elicited through a structured on-line questionnaire. The full questionnaire is reported in Annex A1. The questionnaire was carefully designed

[5] Coral-List is an Internet forum funded by NOAA's Coral Reef Conservation Program. Its purpose is to facilitate discussions on coral reef ecosystems and related subjects. http://www.coralreef.noaa.gov

over a period of three months. Questions were developed from the literature, through discussions with colleagues and the author's own knowledge of literature gaps.

In order not to frame initial thoughts and lead results, the first round of questions were open-ended. Experts were invited to comment on benefits and limitations of implementing PES in a marine setting, the possible use of PES schemes within a wider portfolio of marine conservation instruments and the possible role of PES in coastal development and poverty alleviation.

3.4.3 Questionnaire implementation

During January and early February 2012, the web survey was piloted with six experts specialising in PES systems and marine conservation. Given the wide range of PES schemes possible and the subjective nature of the open-ended questions, unambiguous wording was imperative. Furthermore, given the long length of the survey, design needed to minimised fatigue. Initial piloting identified and corrected these issues to the best ability of the author. A second round of piloting was then undertaken, after which no further changes were made to the survey.

Experts were invited to complete the final on-line survey between February and June 2012.

3.4.4 Response Analysis

Open-ended questionnaire responses were coded using a 'grounded' approach (e.g. Charmaz 2006; Corbin & Strauss 2008; Curnock 2010). The method is an open and iterative process in which statement context and underlying meaning are carefully considered and key issues drawn out. This allowed themes and views to emerge from the raw data itself, rather than fixing responses into preordained categories (Moustakas 1994). The conclusion is a wide variety of responses, some of which were later 'nested' within a common theme.

The following gives an example of the coding process:

Q.1.b. What do you believe are the challenges of bringing PES to the marine environment?

Response: The demand is unclear. Often poor service users and providers pose problems for payments. Use rights are overlapping and boundaries are unclear. The nature of the resource is a dynamic one. Monitoring of the ES is perhaps more complex.

The response raises five issues which were subsequently coded: (1) uncertainty in demand; (2) poor service buyers unable to pay; (3) complex user rights/issues with identification of appropriate service providers; (4) dynamic nature of marine resources and (5) monitoring of PES will be complex/expensive.

(1) and (2) were further nested within a central theme: issues in securing appropriate demand finance.

3.5 Expert profiles

During February and July 2012, 57 web surveys were submitted. Of these fifteen were incomplete and disregarded[6]. Thirty experts targeted via email completed the questionnaire, a response rate of 41% (total sample=74). The final results presented are based on 42 web surveys. Participation experts are listed in Annex A2.

3.5.1 Summary of expert knowledge profile

Relevant expert knowledge and experience is displayed in Table 3.2. As can be seen experience is fairly matched across both fields of interest. Marine conservation experts feature slightly higher within the final sample; 60% of respondents noted practical expertise with marine conservation tools. In comparison 52% recorded experience with PES tools. PES authors comprise 38% of the sample. Table 3.3 indicates expert familiarity with PES implementation and literature.

Table 3.2 Expert knowledge and experience across disciplines (% of final sample, n=42)

	No. of programmes/articles			
	0	***1-4***	***5-9***	***> 10***
Practical experience with PES instruments	48	43	5	5
Practical experience with marine conservation instruments	41	33	17	10
Peer-reviewed publications in PES	62	24	7	7
Participating organisations	Advanced Conservation Strategies; CEMARE, CIFOR, Comunidad y Biodiversidad; Conservation International; Coral Reef Research Foundation; CORDIO; Duke University; Ecosystem Equity; Environmental Defence Fund; Fauna & Flora International; Georgia State University; IIED; Imperial College London; James Cook University; Legal Ray Consultants; London School of Economics; National Fisheries Authority; NOAA; NEF; Stockholm University; Sustain Value; Tetra Tech ARD; The Nature Conservancy; US Environment Protection Agency; UNDP; University of British Colombia; University of KwaZulu-Natal; University of Rhode Island, University of Washington; WCS; World Bank; WRI; WWF; ZSL			

[6] Completed surveys were considered those in which respondents answered all compulsory questions as well as at least 50% of open-ended questions.

Table 3.3 Expert familiarity with PES literature and implementation (% of final sample, n=42)

Overall, how closely do you follow the academic research developments within the PES literature? *(e.g. peer-reviewed journal articles, book chapters)*

Do not follow	*Very occasionally*	*Somewhat*	*Closely*	*Very closely*
0	29	38	21	12

Overall, how familiar would you say you are with applied PES instruments? (e.g. how well do you understand the issues relating to PES implementation in the field)

Not at all familiar	*A little familiarity*	*Average familiarity*	*Good familiarity*	*Very good familiarity*
2	24	21	24	29

3.5.2 Expert views on defining characteristics of PES

Prior to questioning specific to marine PES, we took a more broad interest in those criteria respondents believed must be met in order for a scheme to be considered a true-PES.

For the most part, there exists no formal definition of PES within the literature (Sommerville et al. 2009). However, the most widely accepted interpretation – particularly within the academic literature - defines PES by the following five criteria: (1) a *voluntary* transaction; (2) PES involve a well-defined *environmental service* (or land use likely to secure that service); (3) the service is 'purchased' by at least one *service buyer*; (4) the service is 'provided' by at least one *service provider*; and (5) the payment is *conditional* on service provision (Engel et al. 2008; Sommerville et al. 2009; Wendland et al. 2010; Wunder 2006). However in the last few years the definition has softened and become less restrictive (Shelley 2011). More recently Tacconi (2012) defines PES as "a transparent system for the additional provision of environmental services through conditional payments to voluntary providers".

A wide variety of results were seen, as demonstrated in Table 3.4. The conditionality criteria received the widest consensus, with 81% of participants believing it to be necessary for a true PES scheme. Not mentioned within Wunder's definition, is the need for a PES to be adequately enforced, although this is perhaps inherent in (5). Interestingly, only around a third of interviewees believed PES schemes must be a voluntary transaction, and only 5% that the incentive must be cash.

While it is seen that some experts were previously not well accustomed to PES as an instrument, these results are also believed to be indicative of the growing trend towards a relaxed definition of PES as a tool.

Table 3.4 Expert agreement on those criteria PES instrument must meet in order to be considered TRUE PES scheme

PES criterion	*% agreement*
a. the PES scheme must involve a well-defined environmental service	76
b. the environmental service to be purchased must be purchased by at least one service buyer	64
c. the environmental service to be provided must be provided by at least one service provider	69
d. the parties involved in the PES transaction must be involved in a voluntary capacity	31
e. the PES payment must be conditional on environmental service provision	81
f. the incentive offered for the environmental service must be positive	60
g. the incentive offered for the environmental service must be cash	5
h. the scheme must provide environmental services to a level above those provided in the absence of the programme (i.e. must be additional)	55
i. the environmental service must be provided by service providers with well established property rights	31
j. the PES scheme must be adequately enforced	69
k. none of the above	2

In order to gain consistent results across respondents, PES was defined within the survey as per Sommerville et al. (2009), whereby PES are approaches that (1) transfer positive incentives to environmental service providers that are (2) conditional on the provision of the service.

3.5.3 Expert familiarity with marine PES

Experts were presented with an information page which presented: a definition of marine PES; an example list of possible marine environmental services; and an example of a marine PES vs. a payment scheme not considered a marine PES[7]. Experts were then asked their previous familiarity with the information presented. Respondent familiarity with the information presented and marine PES schemes is displayed in Table 3.5.

[7] The main difference between the two schemes relied upon a conditionality in one scheme vs. a one-off payment.

Table 3.5 Expert familiarity with information presented on marine PES

Were you previously familiar with the information presented on this page?

Not familiar	*Somewhat familiar*	*Familiar*	*Very familiar*
2	33	43	21

Approximately 65% of experts claimed previous familiarity with the statements presented. While just over 35% were somewhat or unfamiliar with the information, it is important to note that experts were elicited due to their experience in PES or marine conservation science and how this then transfers to this lesser known field.

3.6 Results

Tables 3.6- 3.10 present the frequency and distribution of all responses emerging from the data. Italicised topics are nested within overarching themes. However, given the wide range of responses, this paper limits discussion to a number of key topics as well as a few interesting ideas not previously discussed within the literature.

3.6.1 Benefits of bringing PES to marine environment

Table 3.6 displays the full record of the coded results for question 1.a: "*What do you believe are the benefits of bringing PES to the marine environment?"*. As anticipated, experts expressed many of the common and prominent themes pertaining to the advantages of PES more generally, as well as a fewer more specific marine related topics.

3.6.1.1 Behaviour change and the promotion of local enforcement through incentives and compensation

Almost one third of experts mentioned the ability of PES schemes to incentivise behaviour change as an important benefit.

In coastal settings, where poverty and resource dependence are both considered to be high overall, one might expect the provision of compensation for opportunity costs to be an important benefit of marine PES over other previous instruments; indeed the transfer of benefits to resource-users/owners was mentioned by a quarter of experts (28% of responses). However, only five respondents mentioned compensation for opportunity costs specifically. The ability of PES to promote additional – non-financial – benefits was also highlighted (23%). Beyond investment into alternative occupations, securing tenure, empowering local communities, as well as promoting social capital and conflict resolution were mentioned.

Table 3.6 Coded responses: Q1a. What do you believe are the benefits of bringing PES to the marine environment? (n=40)

	No. respondents	***% responses***
Incentivise behaviour change	12	30
Promote local enforcement/lower enforcement costs	5	13
Sustainable finance	4	10
Conditionality of instrument	2	5
Cost effective conservation	2	5
Outcome-based conservation	1	3
Promotes participation/involvement	1	3
Transfer of benefits to resource user/owner	11	28
compensation of opportunity costs	*5*	*13*
monetary incentives	*4*	*10*
transfers ES value to users/owners	*2*	*5*
compensation for time-lag between benefits accruing and closure of fishing sites	*2*	*5*
Highlights value of environmental service	9	23
highlights true value of ES/places value on ES	*9*	*23*
cost of natural capital incorporated into individual/policy decisions	*6*	*15*
Additional social benefits	9	23
investment into alternative occupations	*4*	*10*
local empowerment/ownership of initiative	*2*	*5*
transfer tenure to resource user	*1*	*3*
investment into social capital/conflict resolution	*1*	*3*
protection of fisher livelihoods	*1*	*3*
A need to better protect marine environment	8	20
current poor condition of marine environment	*6*	*13*
failure of existing instruments	*3*	*8*
Improved environmental performance	4	10
improved marine sustainability	*3*	*8*
precise spatial targeting	*1*	*3*
lower incentives required in marine PES (due to faster lifecycle regeneration)	*1*	*3*
reduce fishing effort	*1*	*3*

The ability of PES schemes to promote local enforcement and reduce enforcement costs was highlighted by 13% of experts, and as mentioned by one respondent, may be particularly valuable within a marine setting:

> *"Community conservation has the criticism that it does not ensure environmental protection…, in a large part because "new" profitable activities are integrated as complements rather than substitutes to the "old" environmentally damaging livelihoods. That criticism was primarily for terrestrial conservation and basically*

due to the fact that it's hard to exclude people from e.g. forests. It seems to me marine conservation would have a much more difficult time of such exclusion, so the need for a truly conditional conservation mechanism is greater. You don't have to exclude only monitor…change the incentive structure and monitor well… So where enforcement is very difficult (e.g. coral reef) a conditional mechanism that only required monitoring, and not exclusion, may be less effort and a more effective use of money."

3.6.1.2 PES as a transitory instrument

PES was considered particularly well suited to the marine environment by two experts for characteristics little discussed within the literature. And in stark contrast to the general view of marine PES being a greater challenge.

"PES particularly useful in areas where the spatial resource use pattern is important (its difficult to achieve surgical precision in resource use patterns with ITQs for example)"

"…the key difference is that marine resources tend to regenerate much faster than terrestrial resources. This difference makes marine PES far more likely to be successful at lower levels of incentives than for terrestrial resources. For example, local people will probably see fish stocks recover much quicker after a closure, and this will encourage them to support the PES programme and to support the new institutional framework created (e.g. enforcing closure). With terrestrial systems fundamentally sustainable management is not in the interests of people because the rate resources regenerate is slower than investing the capital in another type of land use. Under these circumstances I think payments need to be larger and sustained for longer"

Indeed, the notion that PES payments, while initially conditional on ES delivery, need not be continuous over the entire period of marine conservation was mentioned by a small number of respondents (n=3). These experts toy with the idea of PES as a bridge to cover short falls in loss of earnings from initial management restrictions. Once stocks recover to a level above initial costs, payments can be weaned out: a transitionary payment as it were. As one expert put it, a 'kickstart' to more sustainable opportunities.

"Potentially a way to offset the opportunity costs of marine management (e.g. MPA or gear restrictions), and encourage participation. This is particularly important in areas where resource users are often poor and, although they often have diversified livelihoods, they are generally limited in economic opportunities. Income from PES could be used to kickstart other more 'sustainable' livelihood opportunities."

"If I believe no-take zones work, then fishers will also see them work, but they need to be compensated for this time lag in giving up areas of fishing grounds."

"Possibly play a key role in getting a system started. Start up costs are a problem in marine conservation and this may be one way to help get over this barrier."

3.6.2 Challenges of bringing PES to marine environment

Perhaps more interesting from a research perspective are those challenges which need to be addressed in order to transition PES to the marine environment. Of the 42 expert surveys submitted 41 responded to Question 1b. *What do you believe are the challenges of bringing PES to the marine environment?* Results are displayed in Table 3.7.

3.6.2.1 Complex tenure in a marine setting

As can be seen, by far the most pertinent challenge for marine PES raised by experts relates to the complex tenure systems seen in marine and coastal areas; as many as 61% commented on this issue. Of these 76% mentioned a lack of property rights more specifically: 46% of the total sample. Multiple users and the identification of appropriate service providers were also cited as an issue, 29% and 27% of respondents respectively.

As put by one expert:

"One particular related problem that is acute in the marine environment is the lack of property rights in space or species and thus the difficulty in enforcing claims."

Additionally;

"Different from forests or terrestrial ecosystems, marine ecosystems are not owned by any well-identified owner. Instead many users gather on these ecosystems and it is therefore difficult to identify one service buyer and one service provider."

3.6.2.2 Marine ecosystems

The nature of marine ecosystems was stated as a challenge to marine PES implementation by over a quarter of respondents. Of these, the dynamic nature of marine ES was cited by a smaller subsample. Closely related to this issue was the dislocation between service provision and end product (n=3). Indeed, as one expert noted, within marine systems organisms are not only highly mobile but can require varying environments at different stages in their life development:

"Marine animals are highly mobile; as adults and at early life stages. Sustainability of fish stocks and biodiversity is dependent on many factors that affect them differently at different life stages."

Table 3.7 Coded responses: Q1b. What do you believe are the challenges of bringing PES to the marine environment? (n=41)

	No. respondents	*% responses*
Complex tenure systems in marine & coastal environment	25	61
lack of tenure/property rights	*19*	*46*
many resource users (sometimes conflicting)	*12*	*29*
difficulty in identifying appropriate service providers/owners	*11*	*27*
complicated contracts	*1*	*2*
Effective monitoring & enforcement	17	42
enforcement difficult and complex	*12*	*29*
monitoring complex and expensive	*9*	*22*
free-riding	*4*	*10*
exclusion of others e.g. roaming bandits/poachers	*3*	*7*
ensuring compliance	*3*	*7*
ensuring conditionality	*1*	*2*
Nature of marine ES	11	27
dynamic nature of marine ES	*4*	*10*
externalities/outside damaging influences	*3*	*7*
unclear boundaries	*3*	*7*
dislocation between production and consumption point	*3*	*7*
spatial connectivity of ES	*2*	*5*
slow recovery of ES	*1*	*2*
temporal nature of life stages of marine environment	*1*	*2*
protection of marine environment requires large scale	*1*	*2*
Lack of scientific knowledge about marine ES	13	32
inadequate scientific knowledge of complex marine ES flows/delivery channels	*7*	*17*
difficulty in valuing marine ES/inadequate scientific knowledge about true marine ES values	*4*	*10*
uncertainty in generating and proving additionality	*1*	*2*
risk undervaluing in scale up	*1*	*2*
inadequate science on how to address/counteract marine threats	*1*	*2*
Uncertainty in available level of demand finance	8	20
unclear demand/who are buyers	*6*	*15*
lack of financial support	*2*	*5*
poor service buyers	*1*	*2*
Negative social consequences	8	20
conflict creation	*4*	*10*
ensuring appropriate distribution of benefits/equity	*4*	*10*
exacerbation of poverty/inequality	*1*	*2*
Difficulties with institutional framework	6	15
lack of regulatory framework	*3*	*7*
government inertia/lack of political will	*2*	*5*
convoluted jurisdictions	*1*	*2*
donor resistance to cash	*1*	*2*
Dislike of PES concept	3	7
commoditisation of ES	*2*	*5*
hinders understanding of real environmental issues	*1*	*2*

Another noted:

> *"Mobility of target resources:…. their mobility across different areas can make it more difficult to yield a positive environmental protection result from protecting a particular habitat"*

Furthermore, one expert questioned whether PES would be able to function at the scale required within a marine setting, an issue mentioned for marine tools more generally.

Worth noting, was expression of the slow positive environmental response (i.e. the reversibility of ES) within the marine environment as a barrier by one expert. For example hard coral reefs can take long periods to regenerate but a short time to destroy. The opposite was previously cited as a benefit (Section 3.6.1.2). This is perhaps not unexpected given the wide range of variation between marine ES overall, and points to the suitability of PES for those marine ES with quicker recovery periods.

The more fluid nature of marine ES over terrestrial ES perhaps gives rise to the subsequent barriers to marine PES as expressed by respondents.

Just over 30% of respondents cited a lack of scientific understanding about marine ES as a barrier. For example:

> *"The invisibility of many (if not all) of the ecosystems provided,"*

> *"Determining the cause and effect relationships, managing situations where there are complex interactions."*

Half of these responses (17% of total sample) made mention of the inadequate scientific knowledge surrounding complex marine ES flows and delivery channels. Somewhat related was a concern that there was a greater difficulty in accurately valuing the true value of many marine ES: as mentioned by a third of this subsample, 10% of the total group. Indeed, difficulty in generating and securing additionality was further mentioned by one expert specifically. Interestingly, concern was expressed about the risk of undervaluing marine ES during the scaling up process.

> *"There is also a major risk of undervaluing the marine ecosystem when case studies are applied at a larger scale… This is because the larger scale the more difficult it is to replace the ecosystem goods and services and interactions are too complex to understand impacts of alternatives."*

Experts also cited externalities as a greater concern than in the terrestrial environment. These externalities challenge both the quantity and quality of the service provision. PES will be only be effective in those areas where it is unaffected by outside forces such as run-off pollution. The most obvious externality faced by the marine ecosystem is perhaps that of rising sea temperatures and coral bleaching.

3.6.2.3 Effective monitoring and enforcement

Linked to issues previously discussed such as complex and dynamic ES, unclear boundaries, complex tenure rights, and in stark contrast to the benefits mentioned in the previous section, effective monitoring and enforcement of marine PES were mentioned as challenges by just under half of the expert pool (42%). Just fewer than thirty percent of all responses collated mentioned that enforcement would be difficult and complex within a marine context. Complex and expensive monitoring was also cited by 22% of respondents. Free riding of ES benefits and difficulties in excluding roaming bandits and poachers were highlighted responses.

3.6.2.4 Securing demand finance

All of these challenges and attributes perhaps feed into to a further challenge: uncertainty in demand. Concerns in securing demand were mentioned by just less than one-fifth of experts. The most prominent concern related to identification and securing of buyers for these more 'invisible' ES (n=6; unclear demand in Table 3.7). Lack of financial support was highlighted by a further two experts. One response highlighted the issue of poor service providers. Indeed coastal fishing communities have been identified within some of the most vulnerable socio-economic groups. Asking such communities to pay for the provision of improved ES would likely be unviable as well ethically unsound.

3.6.2.5 Social implications

Remarkably, no one directly mentioned the injection of money into cash-strapped coastal communities. However a likely knock on from this, negative social consequences was mentioned in 20% of responses. Three effects were coded: conflict creation (10%); difficulties in securing appropriate distribution of benefits (10%); and an exacerbation in inequality (2%).

3.6.3 PES within the marine conservation portfolio

Experts were also asked about the potential of marine PES as a 'stand-alone' instrument. Coded responses are displayed in Table 3.8.

Table 3.8 Coded responses: Q2.a. Do you believe marine PES have the potential to better protect the marine and coastal environment over other marine conservation tools? (n=41)

	No. respondents	***% responses***
YES	11	27
where local conditions enable	*4*	*10*
where barriers overcome	*3*	*7*
direct incentives more powerful	*2*	*5*
incentive structure more scalable	*1*	*2*
NO	5	12
strict enforcement still required/issues of ill-defined tenure	*2*	*5*
unviable for severe degradation &/or large areas	*1*	*2*
does not address fundamental issues of expolitation	*1*	*2*
moral hazard/bad precedent	*1*	*2*
COMPLEMENTARY	27	66
brings additional benefits to portfolio which other instruments are lacking	*25*	*61*
will have difficulties working alone	*5*	*12*

3.6.3.1 Marine PES as a stand-alone tool

Just under 30% of respondents believed marine PES has the potential to better protect the marine and coastal environment over other existing tools. However, almost half of these responses were caveated with the mention of barriers to be overcome or definitive conditions which need to be put in place to enable successful implementation. Only two of the 11 positive responses mentioned actual benefits of marine PES above other instruments. The direct incentive structure of PES was cited as more powerful than other current tools; this direct incentive structure was also believed to be more scalable over the larger seascape than other instruments by further respondent.

Only 12% of respondents answered question 2.a. negatively, doubting the potential of marine PES to better protect the marine and coastal ecosystems. Of these, two respondents cited the issues of enforcement as it related to ill-defined tenure within the seascape. Another claimed PES as unviable for those areas where extreme degradation had occurred, or indeed over large areas more generally, and as such less suited to a marine setting. Indeed, a couple of responses raised issue with the apparent 'cost-efficiency' of PES in the marine environment: a major 'selling point' of PES instruments within the literature. Backed up also by the high proportion of respondents raising issue with monitoring and enforcement in Section 3.6.2.3.

"In practice, I wonder whether all the resources intellectual and otherwise put into the development of the concepts and strategies and policies and implementation etc are not better used elsewhere."

"PES has more costly monitoring and enforcement issues."

Only two respondents felt marine PES an unsuitable instrument in principle. One expert felt that the incentive structure sets a bad precedent within the marine conservation as well as promoting moral hazard. The other did not believe PES would address the root causes of over-exploitation, which it was stated would require a greater focus on individual values and behaviour.

3.6.3.2 Marine PES as a complementary tool

By far the greatest consensus was for the use of PES within a wider portfolio of instruments, implementing PES alongside other marine conservation tools. Sixty one percent of experts believed as a tool, PES could bring additional benefits over other pre-existing mechanisms.

To name a few:

"Conditional incentives will be a good addition to the arsenal of marine conservation."

"PES schemes in terrestrial systems have proved to strengthen institutional alliances…They have also helped reduce migration away from rural areas and maintain traditional methods in resource management. They helped generate and maintain new sustainable economic activities and employment."

"They may complement other means or possibly play a role in getting a system started. Start up costs are a problem in marine conservation and this may be one way to help get over this barrier."

"The benefits of marine PES would be that some businesses would be compensated for reduced levels of effort and could therefore remain in business. This allows communities to maintain their traditions,"

When asked in more detail about the potential complementarity of marine PES (Question 2.b, Annex A1), a resounding 93% of respondents expressed a positive response for the potential of marine PES when used alongside pre-existing instruments.

"I predict that we'll see that hybrid systems that combine regulatory protection, some stakeholder management involvement and PES will be the most successful. Because each can be used to complement and address the others' weaknesses"

One expert went as far as to say that PES should not be viewed in isolation.

"PES should be viewed as a set of best practices that can be employed in virtually any ocean and coastal conservation project that involve right-holders, right holder commitments, and project funding."

Another stated PES would be useless in isolation.

"I think they are useless in isolation. First and single prize for a successful marine conservation strategy will be a MPA under a co-management approach where long-term sustainability (i.e. operating costs and profits) is obtained through a sound scientific and socio-economic PES scheme."

These observations are particularly astute for use within local artisanal fishing communities. Moreover, one expert highlighted the issues of cash-injections into local institutions, and under which these local schemes should perhaps be utilised more as a last option.

"any PES programme has to think very carefully about the impact money has on the local institutions, perceptions and social norms. This isn't well understood or thought about in great detail. I would certainly advocate for more traditional conservation tools first (that don't involve $$), and then resorting to PES if its felt that the payments or incentives from PES are necessary to encourage sustainable resource management."

3.6.4 PES and pro-poor design

By far the greatest disparity of expert opinion centred on the extent to which marine PES, and indeed PES more generally, should address poverty within its design. The overarching themes are displayed in Table 3.9. A number of key underlying elements are also coded and displayed.

As can be seen from Table 3.9, experts expressed wide disagreement as to the extent marine PES should attempt to address poverty issues. Just over 45% of the sample stated that marine PES must address or should largely attempt to address poverty and be pro-poor in design, 21% and 26% respectively. Indeed, as many as one fifth of the expert pool stated that marine PES MUST attempt to address poverty. Of this twenty percent, half cited long-term sustainability as the main motivator. Although legitimacy and compliance were only mentioned once within this overarching theme,

their importance should not be overlooked. Legitimacy and compliance, in particular, although not directly mentioned are expected to be explicit in many other issues such as long-term sustainability. The large degree of overlap of coastal areas and high levels of poverty was further stated as reason for strong support of pro-poor PES design by two the experts.

Table 3.9 Coded responses: 3.a. To what extent should a marine PES scheme explicitly attempt to address poverty and be pro-poor by design? (n=39)

	No. respondents	***% responses***
Must address	8	21
long-term sustainability	*4*	*10*
legitimate	*1*	*3*
compliance/support	*1*	*3*
Should largely attempt to address	10	26
compliance/support	*3*	*8*
overlap of coastal ES and poverty	*2*	*5*
trade-offs must be considered	*2*	*5*
Only when compatible with PES goals	7	20
in areas of high poverty/developing countries	*3*	*8*
complicated enough	*1*	*3*
Should not address	6	15
overemphasis can lead to environmental failure/weaken environmental objectives	*3*	*8*
reduces efficiency	*2*	*5*
not a poverty tool	*2*	*5*
poverty drivers should be addressed at source	*1*	*3*
Inherent in design	3	8
through compensation offered	*2*	*5*
due to voluntary nature	*1*	*3*
Unsure	2	5

One expert went on to discuss implications for the non-participating poor, a topic rarely covered in the literature.

> *"If the scheme is seen as illegitimate, you risk certain community members overfishing the area to be protected or directly sabotaging the programme. But I would also be concerned about the implications for the poor non-participants. There are examples where an influx of money by certain groups cause a massive inflation of staple foods in the local markets, such that the poor could no longer afford those foods."*

In stark contrast 15% of experts indicated that marine PES should not be addressing poverty in its design. Half cited the loss of environmental gains as the predominant

reasoning (8% of total sample). One respondent maintained that PES would have difficulty dealing with the inherent causes of poverty.

> *"There is a problem in making PES pro-poor as the drivers of poverty probably need to be addressed at the source – i.e. more equitable distribution of resource rights etc. It is difficult for PES to reverse engineer what are fundamental injustices at a deeper level."*

Another went as far as to say that poverty should be dealt with as a separate instrument all together.

> *"Solve environmental problems with one instrument, and solve poverty problems with another, and don't worry about the interactions between the two…Don't try to solve two problems with one instrument unless you can definitively show that the two problems are causally linked (which we haven't). Lets do "non-poor" PES and measure the impact on poverty. Perhaps there will be positive effects for the poor that are not directly related to the transfers themselves."*

The same expert went on to say:

> *"Don't screw the early marine PES schemes up with more constraints on their ability to function. Let's figure out how PES works in the marine environment without additional targeting requirements and then we can fiddle with it if we don't like how things are turning out on the social side. Don't assume lack of targeting to poor means poor wont benefit."*

Issues with trade-offs between pro-poor design vs. efficiency and environmental performance were also made by some of those who believed marine PES should attempt to be address poverty to some degree. In response to the question one expert wrote:

> *"This is a reasonable goal, but should be approached with care. Too much social engineering can ruin a reasonably good market-based approach."*

A further subset of experts indicated that marine PES should only address poverty when this is explicit within the goals of the intervention, 20% of the final sample. However of these, just under half, n=3, stated that it should be an explicit goal in low-income/less developed countries.

Interestingly, three respondents believe PES to be in a sense inherently 'pro-poor' in design. Two respondents claimed that marine PES would alleviate poverty through the compensation mechanism. The other stated that the inherently voluntary nature

of a PES scheme would mean that those not benefiting would simply decline to participate.

3.6.5 Essential conditions in bring PES to marine environment

In a final question experts were asked what conditions they felt absolutely essential for a marine PES to function successfully. This question received the lowest response rate of the six questions: 27 of 42 respondents. Results are reported in Table 3.10.

Again enforcement and tenure were predominant responses. Adequate enforcement was an essential requirement in 26% of responses; tenure was cited by 15%.

Interestingly, and not previously highlighted in expert responses is that "good understanding of social and economic implications" was the second most stated condition, five responses: 19%.

Table 3.10 Coded responses: Q.8.a. Are there any conditions that you see as being absolutely essential for a marine PES to function successfully? (n=27)

	No. respondents	*% responses*
Adequate enforcement	7	26
Good understanding of social and economic implications	5	19
Adequate monitoring procedures	4	15
Tenure over ES	4	15
Good institutions/governance	3	11
Stakeholder involvement/participatory process	3	11
Equity/distributional considerations	3	11
Good understanding/science of ES	2	7
Clear/simple ES path	2	7
Appropriate incentives	2	7
Appropriate sanctions	2	7
Use in combination with other marine instruments	2	7
Demonstrate ES value to buyer	2	7
Legitimacy	1	4
Long-term financing	1	4
Compliance	1	4
Transparency	1	4
Education programmes	1	4

3.7 Discussion

Expert elicitation highlighted the benefits and challenges in bringing PES to the marine environment, as well as to investigate how best to transfer these instruments. Various themes were raised by experts across all questions. Many common and prominent topics were mentioned as well as some lesser-cited issues.

While the current literature focuses very much on the opportunities for marine PES, there is little critical analysis of the difficulties of implementing marine PES. For example, how these may differ from terrestrial models, and to what degree these are indeed limitations. For this reason, within the following themes, we focus discussion on the barriers where research is perhaps more pertinent.

3.7.1 Challenges for marine PES

3.7.1.1 Tenure

As expected, issues relating to tenure featured prominently within the data. Indeed, issues pertaining to complex tenure systems within the coastal and marine environment were cited by nearly two thirds of respondents and just fewer than half mentioned a lack of property rights more specifically. In fact, issues of tenure were mentioned by 50% more respondents than the next highest featured limitation.

However, in defining PES criteria only approximately a third believed ES need be provided by service providers with well-established property rights. Within the terrestrial PES literature similar concerns have been voiced, yet PES arrangements have been established and functioning despite land titling not being fully formalised (Vatn 2010). However, under the relaxation of a 'true PES' scheme, the literature speaks of PES as a means to cement property rights for resource owners (Landell-Mills 2002; Muradian et al. 2010). This in itself is seen as a positive incentive. In many coastal and marine settings the state is regularly the rights holder (Lau 2012); communities often have little control over the resources on which they depend, incentivising individual over-exploitation. As such, cementation of property rights could prove an important incentive in itself. In Bolivia the Los Negros pilot PES project helped participants demark land thereby giving it higher de facto protection from landless migrants (Pagiola et al. 2005; Wunder 2008). In a marine setting, the implementation of ITQs within industrial fisheries has had positive economic and environmental consequences (Costello et al. 2008; FAO 2008; Worm et al. 2009). While ITQs prove difficult to implement in artisanal fisheries due to multispecies catch and multiple landing sites PES schemes could hold similar opportunities.

The past few years have seen the advancement of new instruments for ocean governance which transfer property rights to local communities. These include such examples as community-based management (CBM), ocean zoning and marine conservation agreements (Lau 2012). Furthermore, these mechanisms can be highly successful. For example, community-based MPAs have been shown to be effective in achieving conservation and fishery targets (Horigue et al. 2012). And in those areas

with less experience with such instruments, potential exists; in Tanzania, legislation exists which can grant property rights over inshore waters to Beach Management Units (BMUs). BMUs comprise community fishing organisations which are in turn able to set management rules and control access to fishery resources (Tanzanian Fisheries Division 2005). In 1999, Chile passed legislation which grants exclusive territorial user rights for fisheries[8] (TURFs) to registered artisanal fisher organisations within inshore coastal areas (Gelcich et al. 2008).

PES can work alongside pre-existing instruments to help transfer and secure local property rights, and while this will indeed be a challenge this ability should also be seen as an opportunity. As one expert put it:

> *"PES schemes can also act as a catalyst in areas where there is no legal framework behind an ecosystem, which is especially true in marine environments which remain one of the least protected environments in the world."*

Or another:

> *"I do not think investing in clarifying property rights and local management institutions is key. This doesn't necessarily need to be a pre-requisite for establishing a PES scheme, but should go in tandem with efforts to establish PES."*

The main challenge for marine PES as it relates to tenure is perhaps not the lack of it, as mechanisms exist to transfer rights where governments are willing, but in fact the identification of appropriate stakeholders. As mentioned by approximately a third of the expert pool, marine systems are subject to many resource users within a small finite space. Difficulties were cited relating to the identification of appropriate service owners. Indeed creating legitimate marine PES schemes will be a challenge. With so many stakeholders, there will undoubtedly be winners and losers.

Indeed, perhaps one of the most obvious ways forward in coastal PES, at least for those instruments targeting artisanal fishing communities, is application of community contracts. Community contracts target many users, but in doing so include a complex set of incentives to induce participation, i.e. payments, outreach, legal frameworks (Sommerville et al. 2010). However, given that these artisanal communities are some of the most vulnerable worldwide, defining these common property rights and creating equitable and fair incentives will be more relevant.

[8] TURFs allow beneficiaries the right to limit access to fishery resources within a limited sea territory, to determine the amount and kind of resource use and to extract benefits from the use of these resources as well as from future returns (Christy 1982).

However, marine PES can learn from previous community instruments in areas of equity. Sommerville et al. (2010b) look at PES community contracts within forestry communities in Madagascar and make recommendations for improving perceived equity. Work by Fisher et al. (2010) has already linked CBM and PES in fresh water systems – as can be seen in Table 2.2 of the previous chapter. The authors show that lessons learnt from CBM can shed light on key implementation issues in PES. Blom et al. (2010) also show what lessons can be learnt from previously implemented ICDPs. Similar lessons can be learnt within the coastal environment.

A recent paper by Swallow et al. (2009) has made further steps towards identifying and characterising relevant stakeholder in PES schemes, which they label "ecosystems stewards". While such criteria may be in their infancy, creating an infrastructure for the identification of appropriate stakeholders will be key in marine PES implementation and much more work is needed in this area.

3.7.1.2 Enforcement

Related to this theme of multiple resource users and the identification of appropriate stakeholders are issues of enforcement and monitoring. Under these conditions exclusion of inappropriate actors is considered more difficult. Just under a third of experts mentioned effective enforcement as a barrier in transferring PES to the marine environment. Cited as an essential condition for successful marine PES schemes, it is highly unlikely that marine PES will be able to function without adequate enforcement.

However, PES is by no means alone with this issue; this is a story which is widespread across marine conservation interventions. A common tool in low income and artisanal fisheries, MPAs show disappointing results in compliance and enforcement of regulations (Hargreaves-Allen et al. 2011) and the failure of community-based management initiatives is often blamed on ineffective enforcement (Crawford et al. 2004). In fact a recent study by Mora et al. (2006) claimed only 2% of MPAs to be adequately protected. Within the marine environment particularly, local community perception of conservation instrument is an important contributing factor to success (Christie 2004). As one expert put it:

> *"compliance is the biggest challenge in marine conservation,"*

Inducing local compliance may in fact be an advantage of marine PES over other tools. Pollnac et al. (2001) cite "successful alternative income projects" and a relatively high level of community participation in decision making" as two of six key factors in

the overall success of community based MPAs. Perhaps one of the most valuable services a marine PES can provide is to inspire local enforcement and transfer ownership to local communities. As such marine PES schemes can be seen as a mechanism to induce participation, inspire compliance and promote local enforcement.

PES also include a conditionality not previous seen within marine management tools. Including incentives which are conditional on behaviour changes differentiates PES from previous interventions which merely promoted alternatives in isolation (Sievanen et al. 2005; Ferraro & Kiss 2002). In particular, where projects need to induce support from non-participants, benefits can be shared across communities to induce wider compliance. Well-designed in-kind benefits have a greater potential to reach non-participants and reduce conflict. Although not cash, well-designed in-kind incentives should retain conditionality. As always there will be winners and losers, therefore, alongside sustained benefits, enforcement of community contracts should also rely on graduated sanctions.

With multiple resource users, local buy-in will depend on making sure environmental and economic improvements are retained by relevant communities. Yet, while local enforcement and sanctions can induce in-house compliance, other stakeholders do exist. Migrant and roaming fishers as well as overlapping use areas are common within marine environments (Ferse et al. 2010; Daw 2008). As mentioned by a subsample of the expert pool, these parties can create difficulties in guaranteeing benefits both locally and globally. While this can initially be seen as a problem, again it is a widespread problem within marine environments. As discussed before with the cementation of tenure, marine PES can provide infrastructure as well as finance to enable better protection of these environments from outside forces, channelling long-term benefits to local and global stakeholders and promoting sustainable practices.

Even when compliance is strong, leakage can continue to be an issue. MPAs generate spill-over benefits whereby fish stock disperse out from the protected area. However, this is often accompanied by fishers concentrating efforts at reserve boundaries (Kellner et al. 2007). As such, environmental benefits remain localised. Marine PES schemes have the advantage that they rely on incentivising behaviour change. In this way marine PES have the potential to promote more sustainable behaviour and not just displace fishers to MPA boundaries.

3.7.1.3 Monitoring

Monitoring is an important facet of any PES scheme, and if not robust conditionality is lost and incentives are unlikely to influence behaviour (Sommerville et al. 2011). Experts cited complex and expensive monitoring as a challenge to successful marine PES as linked to the nature of marine ES. Furthermore, the lack of scientific knowledge about marine ES was seen as a significant barrier. Worth noting was one concern relating to undervaluing marine ES when scaling up projects given complex interactions, and larger difficulty in replacing ES.

To a much greater extent marine environmental services are dynamic and invisible. This indeed does create a challenge in securing additionality and guaranteeing environmental performance in marine PES. Although more data poor than terrestrial systems, sufficient ecological understanding exists as it relates to how management decisions can improve ES delivery in marine and coastal systems (Lau 2012). For example, no-take zones have been shown to increase fish population and biomass, as well as showing spill over into adjacent areas (Lau 2012; Russ et al. 2003; Williamson et al. 2004). Moreover, mangrove forests are long known to buffer coastal areas from storm activity (Barbier et al. 2008; Dahdouh-Guebas et al. 2005), as well as a more recent acknowledgement of their part in carbon sequestration (Murray et al. 2011; Weaver 2011; Zwick & Kett 2010).

Action-based payments are widespread in terrestrial ecosystems and continue to be more common throughout all project types. These payment types assume relationships between actions and environmental outputs (Skutsch et al. 2011; Sommerville et al. 2009; Wunder et al. 2008). Payments can be made for management inputs and/or opportunity costs incurred (Skutsch et al. 2011). However, it is true that using such proxies may reduce overall efficiency of payments, and moves should be made to pilot PES which are more output-based. However, as in terrestrial, action-based payments can be a good starting ground for marine PES.

Moreover, definite improvements have been made in the modelling of marine ES in recent years. Recently new tools have been designed which can model ecosystem service linkages within marine and coastal environments, such as the InVEST tool created by the Natural Capital Project[9].

Indeed, within expert responses there was a call for more practical experience with marine PES.

[9] Available at: http://www.naturalcapitalproject.org/InVEST.html
For more information see http://www.naturalcapitalproject.org/marine/MarineInVEST_Apr2010.pdf

3.7.1.4 Demand

By far the one of the greatest challenge for marine ES is uncertainty in demand. Although only mentioned by slightly less than one-fifth of experts, without appropriate levels in demand, marine PES will remain unviable. The most prominent issue mentioned by experts related to the identification of buyers.

Previously mentioned challenges limit demand in marine PES. Insecure tenure, diffuse and dynamic marine ES often with dislocated production and consumption points makes identification of appropriate buyers difficult, or indeed makes the case for their buy-in challenging. Securing demand when one cannot identify appropriate buyers nor guarantee service delivery will prove difficult. In addition, few if any working examples of marine PES exist. While some experts speak of marine PES examples, the author knows of no marine PES in practice which are both conditional and financed by private buyers.

Interestingly, no experts mentioned the problem of leakage directly, although it is perhaps inherent in many of the issues raised. Dealing with problems of leakage within a marine environment will be more complex, however insurance mechanisms and confidence buffers can tackle some of these issues.

Promoting buyer confidence is paramount. Assuring buyer confidence will very much depend on previous success stories and improved science surrounding marine ES flows. As discussed in the previous section (Section 3.7.1.3) marine ES modelling continues to improve and new tools are being developed. Improved mapping and trade-off analysis will further assist in identifying appropriate buyers among overlapping beneficiaries (White et al. 2012). Alongside this, marine PES schemes need practical examples. Low-hanging fruits exist. Certain marine PES will be simpler to implement, for example, marine PES which focus on simple ES and small-scale initiatives with few stakeholders. Mangroves and carbon sequestration seem an obvious option. Carbon has pre-developed markets and ES delivery can be easily modelled. But marine PES need to move beyond carbon as well as demonstrate long-term sustainability of finance (Lau 2012; Murray et al. 2011; Weaver 2011).

As in terrestrial PES, it may be the case that government and not-for-profit organisations will take the lead. A growing portfolio of case studies will serve to promote confidence in service delivery and private buy-in.

3.7.2 PES in the marine conservation portfolio

Results showed a clear preference for marine PES to be used alongside pre-existing instruments.

Given the nascent nature of PES, particularly in the marine environment, these instruments are perhaps best to compliment other pre-existing instruments. Indeed, as already mentioned by a number of experts, PES should not be used in isolation. Within the expert pool there was a consensus that PES could address a number of underlying weaknesses in current tools. Sixty percent of experts mentioned bringing additional benefits lacking in other instruments. Marine PES can help to inspire behaviour change and improve compliance through the conditionality criterion. Furthermore, the requirements for investment may promote regional and local development of regulations such as devolution of property rights. For some tools, such as MPAs, marine PES may be a simple tweak to current management policies.

The key question which ultimately leads on from this is then to what extend and under what circumstances should marine PES schemes combine with current policy tools. Again more pilots and working examples of marine PES will help answer such questions.

3.7.3 Marine PES as a pro-poor instrument

By far the widest divergence in opinions was seen in the degree to which marine PES schemes should be pro-poor. One fifth of experts believed marine PES must address poverty, whereas 16% stated that schemes should not attempt to address poverty within design.

Disagreements mainly revolved around the loss of environmental gains in project design (i.e. efficiency) vs. long-term sustainability. For some, the most immediate need was to demonstrate that PES could work in a marine context as previously noted in Section 5.4. Others stated that PES schemes could not address the underlying drivers of poverty and as such should not attempt to do so. For others, however, marine PES schemes which did not seek to address poverty would be seen as illegitimate and fail in the long-term.

With a high degree of overlap in coastal ES and impoverishment the degree to which PES attempt to recognise poverty is an important question – at its very least it should do no harm. With multiple actors and possible community contracts, perhaps the question is not whether it should be designed as pro-poor but how in fact it should be designed to prevent poverty exacerbation.

Just less than ten percent of experts questioned believed PES schemes to be inherently pro-poor by their very design; this was mainly due to their voluntary nature. As put by one respondent:

> *"A marine PES scheme will be pro-poor if it makes poor people better off. To me, this is secured in the voluntariness of the agreement, i.e. poor people will only voluntarily "play ball" if they gain. How much rent is offered to people over and above their estimated provision costs (=pure welfare gains) is a matter of programme design and negotiation power…"*

However, in dealing with artisanal fishing communities for example, community contracts are likely to be a predominant feature. This moves away from the notion of individual voluntary participation; in fact, only 30% of experts stated that parties must be involved in a voluntary capacity.

As communities move into agreements with service buyers, there will be winners and losers. Sommerville et al. (2010) show PES success is related to high levels of perceived fairness of PES payment distribution. However, coastal artisanal fishing communities can possess very unique features. Several forms of exclusion and marginalisation occur in fishing communities, and rent appropriation is common (Béné 2003). How this relates to property appropriation may be of particular concern within areas with currently ill-defined tenure systems.

One expert mentioned the possibility of improving PES design towards poverty factors through utilising basic quotas:

> *"Its possible to make PES more pro-poor by some basic quotas (e.g. in one of our programmes we made sure that a % of households were female-headed households)"*

Badly designed marine PES may have barriers to participation which could further exacerbate poverty; this is something which will very much depend on the eligibility criteria and where the 'bar' is set. For example, a PES targeting one type of fishing gear may have implications for other fishers.

Furthermore, some poorer community members may simply be ineligible to participate from the onset, e.g. are farmers only. Interestingly and perhaps worryingly, very little mention of the implication for non-participants has been made both within elicited responses from experts as well as current literature. Of notable exception was one response by one expert.

"I would be concerned about the implications for the poor non-participants. There are examples where an influx of money by certain groups cause a massive inflation of staple foods in the local markets"

More work is needed in determining the implications for non-participants under community contracts, both in a terrestrial and coastal context where little empirical work exists.

Experts made no mention of cash injections into potentially cash-strapped communities as a challenge in marine PES, although one expert mentioned it later in a discussion relating to pro-poor design. Perhaps this is due to a growing acceptance that incentives need not be cash. Indeed, acknowledging that there will always be non-participating poor within community contracts highlights the need for further non-monetary benefits; such can potentially be beneficial to a wider group of stakeholders. This said, one must not lose the conditionality attached to PES in moving towards in-kind benefits. Designing such incentive structures will be an interesting development in PES research.

Again as mentioned above PES may have better success in meeting poverty goals through working alongside other pre-exising tools.

3.7.4 Marine PES as a 'kickstart'

As mentioned by a few experts, a number of marine ES have the advantage of relatively fast regeneration periods. As such, marine PES have the potential to be used as transitory mechanisms.

Conservation interventions have in the past overlooked social needs which in turn has led to subsequent conflict and disregard for the instrument (Christie 2004). Many compliance issues in marine conservation stem from issues of initial opportunity costs. As mentioned by a few experts, marine PES have the potential to compensate user's opportunity costs for initial restrictions. As ecosystems recover and additional benefits begin to accrue from the environmental services themselves, payments can be stepped back and finally stopped. This quality could be particularly useful in many coastal communities where poverty levels are high and few non-fishing activities exist; in particular alongside such mechanisms as MPAs.

3.7.5 Limitations

It should be noted that this paper focuses very much upon how the pre-mentioned factors pertain to artisanal fishing communities. Marine and coastal environments are complex and diverse and of course one model will never fit all. Some feature of a

marine environment will never be amenable to marine PES. As described by one expert, the marine environment can show slow recovery periods but require only a short time to destroy. Interestingly, the opposite was previously cited as a benefit. This is perhaps not unexpected given the wide range of variation between marine ES overall, and points to the suitability of PES for those marine ES with quicker recovery periods. In addition, artisanal fishing communities show high levels of diversity both within and between communities (Coulthard 2012). Different cultural values and understandings of social well-being may result in PES being wholly unsuitable in some contexts.

One should also mention that due to a relatively small sample size, finding consensus within such a wide range of issues is difficult, and given time and energy constraints experts are expected to mention only those which they see as most important.

3.8 Conclusion

This paper adds to the emerging marine PES literature, moving beyond policy pieces to critically analyse expert opinion as it relates to marine PES. More specifically expert opinions are qualitatively analysed as they relate to benefits and barriers of marine PES implementation, their role within the marine conservation portfolio as well as those criteria essential for their successful application.

Given the large, expansive nature of marine resources, expensive enforcement and a number of various users compliance is key, as one expert put it. Legitimate and equitable contracting to the appropriate stakeholders will be required to reduce perverse incentives, free-riding and disregard for marine PES. It is likely then that, at the artisanal level at least, marine PES will rely on community contracting, designed to induce participation and compliance in order to reduce excessively expensive enforcement costs. Marine PES, can in fact, help induce community management and problems associated around collective action, investing in and promoting local institutions.

Investment into generating these local institutions and ultimately securing compliance as well as exhibiting successful case studies will be key to securing demand and ultimately PES success. However, in balancing the needs of artisanal users, it is important that marine PES do not lose sight of their environmental goals and become yet another ineffective conservation tool.

Results indicate that barriers still present themselves for marine PES. However, lessons can be learnt from the terrestrial experiences with PES and from the vast body

of experience in issues of marine management, and to some degree barriers can be overcome. In many cases these are challenges throughout marine conservation instruments and, in actual fact, PES can promote research and legislation relating to such issues as tenure and ES modelling. As such, marine PES should be seen as another instrument in the conservation toolkit, and one which can work alongside and improve pre-existing interventions through the provision of long-needed incentives and conditionality.

Chapter 4

Determinants for participation: a review

4.1 Overview

In this chapter we review the literature surrounding adoption decisions in a number of related and representative fields including conservation agriculture, agroforestry, microfinance and Community-based Management. Household and individual determinants influencing participation decisions are discussed in Section 4.3. Where relevant we also look to fishery exit and compliance decisions within the fisheries literature in order to better interpret fisher decision-making processes. The chapter concludes with hypotheses as to how determinants will affect fishers' response to the proposed hypothetical marine PES in Section 4.4.

4.2 Introduction

It is highly probable that marine PES schemes will rely on contracts made with 'fishing communities', whether that be a fishing fleet, a specific fishery, or an artisanal community. There are two reasons for this assumption:

1. Given the common pool nature of marine resources, contracts are unlikely to be targeted at the individual level; and
2. Devolving tenure or fishing rights will occur at the fishery/community level.

This will have pros and cons for design. Marine PES can be seen as a potential solution to the collective action problem of marine conservation. Fishing communities, whether they be a fishing fleet or artisanal community, can sign up to participate in marine PES schemes. However, participation will require compliance by all fishers with access rights within the target area; for this reason community contracts will be key. As a result any marine PES programme, while voluntary at a community level, may be obligatory at the individual level.

As a result, realising those determinants of marine PES participation is pertinent. In the first instance, *under a democratic system,* community enrolment in a scheme will only take place once a threshold of willing participants has been reached. In the second, under the assumption of utility maximisation, those stakeholders not wishing to participate consider themselves to be worse off under such a scheme. Both are of concern for different reasons. If marine PES cannot inspire enrolment, they are dead in the water. However, perhaps more worryingly, are the implications for those

reluctant to enter into these schemes. Can those PES schemes which make certain members of the community worse-off be justified?

If marine PES schemes are to successfully enter into the marine conservation portfolio they will need to promote participation as well as 'do no harm'.

Marine PES programmes will therefore benefit from an understanding of the following:

1. Design factors which promote marine PES adoption.
2. Determinants of fisher's willingness to participate at the individual level.

Understanding how schemes can be designed to inspire fisher interest has obvious benefits: promoting adoption in order to reach a participation threshold. Instrument design can be extremely important in achieving adequate acceptance and compliance within the fishery sector. Fishers have long been documented to hold varying preferences for conservation management restrictions (Cinner et al. 2009; McClanahan & Mangi 2004) and any restrictions will likely be viewed as unfavourable and therefore not readily accepted (Kabii and Horwitz, 2006). As has been documented, gains and losses are not necessarily valued the same; the loss of a 'bundle' can be perceived as a greater cost than the gain of an equivalent (Kahneman et al., 1990). As such, in order to inspire compliance, instrument design will be particularly important in these settings as well as in rural low-income areas where monitoring and enforcement efforts are often ineffective (Lundquist & Granek 2005; McClanahan et al. 2005; Christie 2004). Identification of restriction trade-offs and an appropriate method for design and analysis are discussed further in Chapter 7.

The second, less tangible, research objective seeks to understand those potential equity issues associated with a non-voluntary PES scheme. Who are those fishers who consider participation a welfare-loss within the community? What are the endogenous barriers to their participation? And perhaps more pertinently, could these be the poorer and more vulnerable members?

In order to answer these questions we require an understanding of who are those willing to sign up and how these fisher's differ from those more resistant community members. The PES literature centres on the assumption that participants will enrol if and when PES payments are larger than the opportunities costs forgone, however the picture is never this simple (Kosoy et al., 2008; Mahanty et al., 2013; Wunder, 2008). To date PES research has largely focused upon ecological, economic and political barriers. Such emphasis is perhaps not surprising given the mandate of PES to

improve environmental outcomes or its reliance upon market forces to achieve this (Petheram and Campbell, 2010). A large part of this literature discusses the capacity of PES to better enable the participation of poorer households through project design and eligibility rules (e.g. Wunder 2008; Pagiola et al. 2005, 2008; Zilberman et al. 2008). Less documented is a critical analysis and quantification of how household determinants influence peoples' perceptions and decisions; in particular how these variables drive or obstruct a change from current behaviours and the adoption of new PES schemes, or indeed how these relate to current livelihood strategies (Petheram and Campbell, 2010; Sesabo and Tol, 2005; Zanetell and Knuth, 2004). In order to design more successful development-conservation programmes, there is a need to better understand those factors which motivate human behaviour and how these relate to the adoption of new and novel livelihood schemes.

4.3 Determinants of participation: a review

The decision of an individual to participate is generally assumed to follow the random utility model (1)

$$U(X_i) = V(a(X_i), b(X_i)) + \epsilon_i \tag{1}$$

whereby X_i is a vector of attributes that characterise the individual (fisher) *i*, $a(X_i)$ is the fisher's profit *i*, $b(X_i)$ is the non-monetary utility of the fisher i, and ε_i is the error term. As such, the utility of a fisher choosing to participate will be:

$$U(X_i) = V(a(X_i) + P, b(X_i)) + \epsilon_i \tag{2}$$

where *P* is the PES payment. A fisher will choose to adopt the PES programme ($Y = 1$) if he does not suffer a utility cost e.g.:

$$U(X_i, Y = 0) \leq U(X_i, Y = 1) \tag{3}$$

$$V(a(X_i), b(X_i)) + \epsilon_i \leq V(a(X_i) + P, b(X_i)) + \epsilon_i \tag{4}$$

In the context of PES, few attempts have been made to understand these attributes (X_i) which drive participation (Kosoy et al., 2008), and none within the marine context. More generally, however, the importance of factors which determine participation in environmental protection programmes have been widely acknowledged; this has led to a considerable amount of research on schemes such as the UK's Agri-Environmental Schemes (AES), the US's Conservation Reserve Program, forestry conservation projects more generally and CBM initiatives (Cooper

2003; Defrancesco et al. 2008; Falconer 2000; Knowler & Bradshaw 2007; Mullan & Kontoleon 2009; Wossink & van Wenum 2003; Vanslembrouck et al. 2005).

More recently however, the participation literature has expanded to encompass PES scheme determinants (Zanetell and Knuth, 2004). However, with the exception of a few works (see Chen et al., 2009; Mullan and Kontoleon, 2009; Pagiola et al., 2008; Uchida et al., 2007; Zbinden and Lee, 2005), PES studies have tended to focus on a developed world context. There is presently limited empirical evidence on individual determinants of PES participation within the developing countries (Sesabo & Tol 2005; Zanetell & Knuth 2004). The study of PES or PES-like participation in a developing country and/or marine context is important because there are a number of reasons why individuals may respond differently. Unlike in a developed context where schemes take place in reasonably well-functioning markets, households are faced with imperfect markets and institutions. Not only this but they face additional constraints such as low and erratic income, difficulties in accessing credit and insecure tenure (Jacoby and Skoufias, 1997; Mullan and Kontoleon, 2009). This is particularly true within the coastal environment where rural fishing communities face limited markets, are subject to high variation in day-to-day catch and income, commonly experience economic reversals and more often than not operate in what is effectively 'open-access' areas (Béné et al., 2010; FAO, 2001; Pollnac, 1991).

Furthermore, scant empirical research exists on the motivations of individuals to partake in more restrictive programmes for nature conservation (Kabii and Horwitz, 2006); what will effectively be the norm within marine PES. Within low-income countries, participation studies have generally focused upon the adoption of more environmentally-sound or climate-resistant agricultural practices and participation within micro-finance initiatives (Akoten et al., 2006; Diagne, 1999; Nguyen, 2006; Shete and Garcia, 2011; Zaman, 2004) as well as conservation management programmes (Datta and Sarkar, 2010; Musyoki et al., 2013; Zanetell and Knuth, 2004).

Adoption decisions associated with these practices hold much in common with current PES schemes. PES schemes assume a behavioural change. In some PES programmes this will be a setting aside of land or an amendment to current management practices and, as such, an amendment to the individual's current production function. It has been well documented that within imperfect markets household demographics and asset endowment can significantly affect production decisions, both terrestrially (Dercon & Christiaensen 2011; Bandiera & Rasul 2006; Mendola 2005; Van Dusen & Taylor 2005; Smale et al. 2001; Dercon 1998), and in a coastal context (Sesabo & Tol 2005; Allison & Ellis 2001).

Studies on conservation agriculture, agroforestry, microfinance and CBM have identified a number of household and individual determinants influencing adoption decisions. Individual characteristics, attitudes, household assets and structural factors (such as farm structure) are shown to be important (Adhikari and Boag, 2013; Defrancesco et al., 2008) as are perceptions of risk and the benefits of the programme itself (Kabii and Horwitz, 2006). Here we examine the literature as it relates to these schemes, both within a developing and developed context in order to draw conclusions as to how it might relate within a marine PES framework. In addition, where relevant we look to similar adoption, fishery exit and compliance decisions within the fisheries literature in order to better interpret fisher decision-making processes.

4.3.1 Individual characteristics as determinants of participation

Age, gender and education have all been shown to influence adoption decisions.

4.3.1.1 Age, education and household size

Age and education have both been regularly assessed in participation models, but are difficult to link to participation decisions (Knowler and Bradshaw, 2007).

In a recent review of the motivations and determinants influencing participation in land-restricting conservation practices, older landholders were presented as significantly more resistant to uptake (Kabii and Horwitz, 2006). Older landholders were characterised as more sceptical to the benefits of adoption, since benefits may not occur in their lifetime. Moreover, landowners with longer tenure showed greater opposition to restrictive management as they believed their lengthy experience gave them a better ability to deal with conservation threats (Kabii and Horwitz, 2006). However, overall studies show varying results for age as a determinant in the adoption of conservation agriculture. Age has demonstrated positive, negative and insignificant correlations in uptake decisions across a number of studies (Clay et al. 1998; Knowler & Bradshaw 2007; Mercer 2004; Neill & Lee 2001; Okoye 1998). Relating to the adoption of PES, Chen et al. (2009) showed age to be a positive determinant of re-enrolment.

Education is commonly shown to positively correlate with the decision to adopt or participate in conservation agriculture (Deressa et al. 2009; Knowler & Bradshaw 2007; Mercer 2004; Swinton & Quiroz 2003; Traore et al. 1998) as well as PES schemes (Adhikari & Boag 2013; Zbinden & Lee 2005). And higher education is generally found to be positively associated with environmental concern (Olli et al., 2001). However some studies have found education level to be an insignificant determinant

(Clay et al., 1998; Knowler and Bradshaw, 2007) or even negative (Nyangena, 2008). Studies relating to the adoption of microcredit have also shown mixed results: Khandker (2005) found education to have a negative effect on borrowing, whereas Evans et al. (1999) list the lack of education as an important barrier to participation.

Perhaps confounding the lack of significance of these variables as a determinant is the possible correlation between education and age with other variables such as wealth. For example, a study by Defrancesco et al. (2008) demonstrated a greater refusal to enter AES schemes by market-orientated farms which were operated by highly educated and relatively young farmers, who were planning to invest further into their farm business. More generally though, low variation in education among low-income respondents in many studies and heterogeneous data categorisations further compounds comparison of these variables across studies (Baumgart-Getz et al., 2012; Mercer, 2004).

For fishers, it is more common to witness a resistance to change from older, more experienced fishers. These older fishers generally feel there are few other options available to them, and/or it is too late to change (Barr and Mourato, 2009). Baticados' (2004) study of a fishing cooperative in Capiz, of the Philippines, demonstrated age as a negative predictor of willingness to participate in a coastal resource management. Although not age per se but often highly correlated, fisher experience has also been shown to predict fishery exit decisions. In the South China Sea biogeographic region of the Philippines, Muallil et al. (2011) confirm fisher's experience to strongly and negatively determine willingness to exit the fishery. However, not all studies agree; although negative in its direction, Gelcich et al. (2009) could find no significant relationship between willingness of artisanal fishers to participate in the creation and administration of a local MPA in Chile. Although not PES related but perhaps of relevance, age has also been seen to be a determinant of non-compliance with fishing regulations such as illegal gear use and effort-limiting regulations (Akpalu, 2011a, 2011b). However, in this case a younger age often predicts higher non-compliance. Muallil et al. (2011) also found educational attainment to influence the likelihood of fishery exit; a similar study by Cinner et al. (2009) showed no such relationship.

We predict fisher's age to influence willingness to participate within the proposed PES. It is anticipated that this effect will be negative, with older fishers more resistant to change. However, it is possible that given the 'rule-breaking behaviour' of younger fishers these results could be confounded. Education is expected to positively influence adoption decisions, however again given the lack of variability seen within the local context it is not unexpected if no effect is seen.

Like both education and age, household size is also thought to influence adoption and participation decisions. The hypothesis is that increasing household labour leads to an increase in the available labour needed to undertake new adoption behaviour (Nkamleu and Manyong, 2005). However, results are scant and variable across studies. Dolisca et al. (2006) suggest a positive significant relationship between household size and forest management participation in Haiti. Deressa et al. (2009), on the other hand, show no such relationship within adoption decision for climate change adaptation in Ethiopia. Nkamleu and Manyong (2005) show household size to be a significant and positive determinant in only some certain adoption decisions such as adoption of live fencing and apiculture; however this result did not apply to all adoption decisions, nor did it affect only those adoption decisions requiring increased labour. The fishery exit literature offers no insights on the role of household size in determining an exit decision. One might anticipate, that household size could influence marine PES adoption in one of two ways and in combination with other attributes. For example, those with larger households might be able to rely on other income means, or alternatively might have a higher number of people dependent on fishing gains.

4.3.1.2 Gender

Gender is an important attribute for participation and refers to socially pre-determined ideas and practices of what it means to be male or female (Baden and Reeves, 2000; Mercer, 2004). Yet, gender is likely to influence participation indirectly and in a convoluted manner. A female's choice to participate may be restricted by gender norms which can separate male and female roles within rural areas (Agarwal 2009; Agarwal 2001; Mwangi et al. 2011; Sturmheit 1990). These social and cultural norms can influence female access to resources, the resource involved, their control over the resource as well as further access to technology and markets. Furthermore differing educational limitations and household commitments mean that women and men respond differently to development opportunities (Allison and Ellis, 2001; The World Bank et al., 2009). Rural women rarely have legal or – in the case of many coastal areas – *defacto* control over natural resources: women own less than 2% of titled land globally (OECD, 2001). Moreover, women often have limited access to new technologies and skills as well as the education, knowledge and/or confidence to implement them, or indeed the finances to purchase them in the first instance. Nor do women often have the time to invest in their development. Within artisanal coastal communities a woman's identity is very often closely tied to reproductive and household work (De Silva, 2011). For example, women spend considerable amounts of time caring for children and the elderly, preparing meals and collecting water.

These household commitments mean that women generally have much more diverse livelihood strategies than their male counterparts, and a primary purpose to provide subsistence to their household. As such, considerable restraints are placed on what women can realistically do in terms of time, labour and activities outside of the house (Tindall and Holvoet, 2008; Weeratunge et al., 2010).

Female participation has been shown as a significant factor in many conservation initiatives (Agarwal, 2009; Westermann et al., 2005). Within a fisheries context, female involvement has been associated with greater community acceptance of fishery management regulations, as well as reduced conflict amongst fishers (Sultana and Thompson, 2008). An understanding of the differing motivations, norms, capabilities, and incentives is important in facilitating female participation and is largely ignored within the development and resource management literature. Of notable exception are the works of Agarwal (e.g. Agarwal 2001; Agarwal 2000; Agarwal 1997) and more recently, within the realm of artisanal fisheries (Peterson & Stead 2011).

When eligible and able to participate, evidence suggests that women hold different attitudes to their male counterparts. Women have been shown to view natural resources (and fisheries) as a means of meeting basic needs as well as a support mechanism for improving self-reliance; on the other hand males can often view such resources as a source of income (Dolisca et al., 2006; Porter and Mbezi, 2010; Walmsey et al., 2006; Weeratunge et al., 2010). Some claim that this manner in which women view their natural resources make them more likely to promote conservation values and practices (Westermann et al., 2005).

Overall, the effect of gender on adoption is mixed. A recent review of agroforestry adoption studies found gender to be a significant determinant in 60% of explanatory models in which it was included, where the gender variable was described as the proportion of males in the household (Pattanayak et al., 2003). However, it is not clear if this is a 'gender effect' per se or a reflection of the resources available to the household. A study by Nkamleu & Manyong (2005) in Cameroon found male-headed farming households to more commonly adopt agroforestry practices. Wilson (1996), however, found no differences in conservation-orientated attitudes or participation in AES schemes based on gender. However, the author goes on to note that only 8% of all respondents were female and most women owned small farms where profit maximisation was considered an important factor, and possibly over-riding other preferences. Moreover, in contrast, Gladwin et al. (2002) found women in female-headed households were significantly more likely to adopt improved fallows than both men or women from male-headed households. In a similar vein, Chen et al.

(2009) also found females 30% more likely to re-enrol plots in the Chinese SLCP PES programme.

In the marine setting, Peterson & Stead (2011) report gender preferences within an MPA non-compliance study on the island of Rodrigues, West Indian Ocean. The authors report that primary drivers of non-compliance were lack of food and income security. They go on to say that willingness to change occupation may come to be an important factor for compliance in those MPAs with high levels of displacement. Findings show that different groups were more willing to change jobs, and while this binary choice may not be gender related per se, the type of job individuals were willing to consider showed a significant relationship with gender.

This said, gender should not be over-simplified or over-generalised. It is important to note that women, much like fishers, are not a homogenous unit. Varying personal and household attributes will influence female behaviour in a similar manner to male counterparts: income, education, marital status and age are all influencing factors (Nuggehalli and Prokopy, 2009).

Evidence seems to suggest that women hold values which make them more likely to promote environmental protection (Agarwal, 2000; Agrawal et al., 2006; Westermann et al., 2005). However, at the same time women have additional limitations on their time, their technical capacity and their social weave which may make it more difficult to engage with conservation programmes on the same level as their male counterparts (Agarwal, 2000). The general absence of literature on female fishers, or indeed female farmers, makes it difficult to predict the influence of gender on PES adoption. However, within the proposed marine PES scheme the dominant fishing type for women, tandilo, would become illegal. It is anticipated that the initial utility loss experienced by women would for this reason be greater than for their male counterparts. One might assume that the more restrictive nature of the PES restrictions on women may reduce their willingness to adopt the scheme.

4.3.1.3 Income

The literature pertaining to income and adoption of conservation practices can be split into two parts; income can have both a positive and negative effect for two very different reasons. For many adoption schemes, an initial investment is required; here, income generally shows a positive influence on participation for the simple reason that non-adopters cannot afford this initial investment. Indeed, income and farm profit are often seen as a significant and positive determinants in many conservation agriculture practices (Deressa et al., 2009; Knowler and Bradshaw, 2007).

On the flip side of this coin, higher earners may be less keen to adopt new practices for fear of greater losses. Within the fishing literature, willingness to exit fisheries is more often than not, negatively correlated with income (Cinner et al., 2009; Daw et al., 2012; Ikiara and Odink, 2000).

The technology adoption literature here perhaps gives a less accurate indication than it does for other attributes. One of the key features of a PES scheme is that it is able to compensate for loss of earnings. In addition, the restrictive nature of the marine PES does not necessarily require initial investments into additional technologies. Looking to the PES and AES literature may be more appropriate.

Results from PES adoption literature are mixed. Delacote et al., (undated) demonstrate household income to be a positive adoption determinant in the Natura 2000 forest biodiversity conservation programme, a French government incentive where participants can gain exoneration from a "land value tax". A similar result is seen by Baumgart-Getz et al., (2012) who suggest that the burden of investment into better management practices is less severe for these individuals. Langpap (2004), however, find the income as insignificant in an analysis of landholder participation in the US Endangered Species Act.

The variety seen could also be a by-product of the various different types of income which are often lumped as one variable. For instance 'income' can represent on- and off-farm product. The positive influence of 'off-farm' income on PES adoption has been frequently cited and is discussed further in Section 4.3.3.2. The evidence for 'on-site' income is mixed: on the one hand, Deressa et al. (2009) show on-farm in agricultural adoption decisions; on the other, Bergseng and Vatn (2009) indicate on-site income to have a negative impact on participation in biodiversity conservation.

If income is a barrier to the adoption of practices that require change which are not originally compensated for, poorer individuals may be less likely show an interest in the PES (Mahanty et al., 2013; Wunder, 2008; Zilberman et al., 2008).

Overall, within the proposed marine PES one might expect higher, more invested, earners to be more resistant to change and increased restrictions. However, if poorer fishers are also more hesitant to change, or require changes they see as 'expensive', income may show a u-shape distribution, whereby both poorer and richer fishers are less keen to adopt said scheme.

4.3.1.4 Fishing type

It is worth noting that not all fishers are equal. Fishers are a widely heterogeneous group (Béné, 2003), but can be more closely grouped based on fishing type (Crona and Bodin, 2006), e.g. the types of fishing gear used.

Not covered within the participation literature for obvious reasons, these differences may also have implications for PES adoption. For example, Akpalu (2011a) note that fisher compliance with effort-limiting restrictions can be influenced by their skill set. The author predicted that less skilful fishers were more likely to violate fishing regulations which restricted effort. However, overall, little empirical work has looked into the implications of gear type on fishery compliance or exit decisions.

It is anticipated that some fishers may be more easily able to adapt to any introduced PES restrictions, for example those currently using illegal fishing gear will be more heavily impacted hence more resistant to adopt. However, this may be overridden by adequate compensation. In addition, some gears require a greater initial investment and may be correlated with issues of investment and ownership. And akin to those with higher asset investment (as discussed in Section 4.3.3.1) may be less willing to change. As yet, there is no clear mechanism as to how gear type will influence marine PES adoption.

4.3.2 Individual environmental beliefs and attitudes

More generally, attitudes and perceptions are important drivers of behaviour (Falconer, 2000). A resource owner's attitudes have been shown to significantly determine participation choice (Falconer 2000; Wilson 1996). Research indicates that the presence of conservation attitudes is, more commonly than not, a positive motivator (Knowler and Bradshaw, 2007; Langpap, 2004). In fact, an awareness of, and concern for, environmental issues has been cited as perhaps one of the more critical factors affecting adoption (Knowler and Bradshaw, 2007). However, awareness of environmental damages can also act as a negative determinant. Zanetell & Knuth (2004) present findings which suggest that fishers with a high level of concern about the current and future state of their fisheries may be less willingness to participate in CBM programmes. The authors suggest this is due to defeatist attitudes which arise from perceived over-whelming and insurmountable barriers.

Moreover, environmental concern as a significant motivator may take a backseat to economic determinants in those instances where a landholder's basic economic and survival needs are not met, or only adequately met (Zanetell & Knuth 2004).

It is anticipated that those fishers with greater environmental concerns would be more willing to partake in the proposed marine PES with these fishers anticipating greater benefits. That said, one might predict these determinants to show a lesser correlation where fishers are confined by other more immediate motivators such as income and consumption smoothing.

4.3.3 Individual perceptions of risk and vulnerability

Household and individual decisions are not made solely on the financial reward offered. While potential income will affect the choice to participate in new activities, considerations about the riskiness of said action will also play in an individual's mind (Allison and Ellis, 2001; Frewer, 1999; Sesabo and Tol, 2005). Perceptions of risk will be determined by ones potential vulnerability and resilience to income shocks (Ezemenari et al., 2002). The notion that perceptions of risk and uncertainty have important implications for adoption decisions is acknowledged within the participation literature, and has even been demonstrated to increase the required taking-price (Isik and Khanna, 2003). The inability of households to smooth consumption across negative shocks – that is a household's vulnerability – has been shown to lead to underinvestment in profitable but potentially more risky projects (Dercon & Christiaensen 2011; Pearlman 2012) . Risk and uncertainty have long been recognised to reduce the adoption of a variety of agricultural innovations (Mercer, 2004). In practice, households with a lower ability to spread risk may choose to adopt the 'least risky' strategy or maintain the status quo, often perceived as the safest strategy (Pagiola et al., 2005; Tschakert, 2007). Worryingly, it is often poorer households who are more vulnerable and less able to spread risk.

Households have developed a number of mechanisms to mitigate economic and consumptive shocks, particularly within artisanal fishing communities which are defined by high levels of variability in catch and income (Dercon 2002; Pollnac 1991). Asset endowment, occupational diversity, as well as social capital – e.g. local institutions, shared knowledge and norms – are some of the many mechanisms households can employ to spread risk (Alderman and Paxson, 1992; Sesabo and Tol, 2005). Each is discussed below, in turn.

4.3.3.1 Asset Endowments

The accumulation of assets is one adaptation used to manage risk and smooth consumption (Moser 2008, 1998). Asset endowment has been shown to determine the degree to which one discounts future gains. Those who possess more endowments have a greater ability to survive in times of food insecurity (Nyangena, 2008) and this

capacity enables households to invest in unproven and potentially riskier technologies or initiatives but which may be, in the long-term, more profitable (Dercon & Christiaensen 2011).

Asset endowment has received attention within the participation literature in the form of farm size, livestock, savings and tenure as well as ownership of other material goods. Within the agroforestry forum, both the theoretical and empirical literature indicates resource and asset endowments as critical to adoption decisions, where early adopters tend to be wealthier households (Mercer, 2004). In reviewing the recent agroforestry adoption literature, Mercer (2004) cites asset endowment as a positive determinant in all studies where 'assets' were included as an independent variable. Participation in community forest conservation has also been shown to vary depending on crop land, livestock endowment, farm-size and land ownership; where all these factors positively predict higher involvement (Musyoki et al., 2013). When considering microfinance participation, assets such as home and telephone ownership have also been shown to positively affect adoption decisions (Pearlman, 2012).

Secure tenure can also strongly predict improved conservation practices (Gebremedhin & Swinton 2003). In a recent agroforestry adoption review comprising 37 empirical studies, Mercer (2004) found that when significant more secure tenure always predicted adoption, and in only a few studies was it insignificant. No studies displayed reverse correlation. Similar results were shown by Baumgart-Getz et al. (2012) and Pattanayak et al. (2003). On the flip side of the coin, in those areas where insecure tenure is a particular issue, solidification of tenure can be a motivating factor, as has been seen in Joint Forestry Management (JFM) programmes (Datta and Sarkar, 2010).

Farm size is a commonly assessed factor within the adoption literature, and can be particularly important in those schemes which require set-aside or decreased harvesting. One assumes that those farmers with larger holdings will be more able to invest in those schemes which require initial restrictions. However, empirical evidence is mixed (Baumgart-Getz et al., 2012; Knowler and Bradshaw, 2007; Mercer, 2004), and has even shown negative results (Nyangena, 2008). These conflicting results are perhaps due to the conflicting motivations; while larger holdings can represent a greater area with which to set-aside land for conservation practices it can also represent a greater investment into the sector. However, a meta-analysis of AES participation in the US found 'capital' – a measure of investment into the farm controlled for farm size – to be a positive determinant of adoption (Baumgart-Getz et al., 2012)

Asset endowment has received some attention within the PES participation literature. Notably, Zbinden & Lee (2005) show farm size to be a significant positive determinant of participation in Costa Rica's PSA PES programme. Uchida et al. (2007) also found landholdings to be significantly higher for those participating in the Chinese SLCP conservation programme when compared to non-participants. In addition, Ma et al. (2010) identified land area and on-site farming practices as important considerations for PES enrolment. A study by Sesabo & Tol (2005) show similar considerations were present within a coastal setting. The decision for fishing households' decision to participate in various new income-generating activities was influenced by endowments such as land and fishing assets. Determinants of resistance to exit fisheries have been shown to include ownership of fishing assets; greater ownership of fishing assets can predict a decreased willingness to exit the fishery (Cinner et al. 2009; Ikiara & Odink 2000). Moreover, Barr and Mourato (2009) found that fishers' owning their own equipment would require larger incentives to sign up a marine PES which restricted access to previously fished sites.

Interestingly there are perhaps two types of asset one must consider within marine PES, as presented within the fisheries' exit literature and PES literature more generally. Although assets can spread risk, they also can represent a greater investment into the sector. For this reason one might consider fishing assets and other assets as having different and opposite influences on marine PES adoption. It is predicted that fishing assets will negatively influence adoption and other household assets will have a positive effect.

4.3.3.2 Occupational diversity

Income and occupational diversity are other adaptions to smooth consumption and manage risk within developing countries (Alderman & Paxson 1992; Dercon 2002; Reardon et al. 2007), and an important strategy within many fishing villages, where fishers commonly involve themselves in other economic sectors to smooth the effects of catch variation (Allison and Ellis, 2001; Coate and Ravallion, 1993). Quite simply, a multiple income portfolio reduces the risk of livelihood failure by spreading it across more than one source thus improving the ability to withstand shocks (Allison and Ellis, 2001). In addition, experience with other activities may give possible actors a greater confidence of success in undertaking new activities, as has been shown in the AES literature by the likes of Baumgart-Getz et al. (2012), D'Emden et al. (2008), D'Souza et al. (1993) and Defrancesco et al. (2008). Interestingly, results from Kosoy et al. (2008) indicate that the diversification of production activities through, say, PES schemes can act as a participation driver.

Terrestrially, off-farm activities have been shown as an important determinant in a number of agri-conservation studies, as well as PES (Chen et al., 2009; Deressa et al., 2009; Knowler and Bradshaw, 2007; Zbinden and Lee, 2005). Those with investment into alternative occupations are believed to have proportionally lower opportunity costs associated with decreased production, are less concerned about possible hidden costs of programme adoption and are more interested in potential on-site amenities generated by conserved areas (Kabii and Horwitz, 2006). A study of participation within the Costa Rica's national PES programme, found participants had higher incomes and were proportionally more reliant on off-farm sources when compared with non-participants (Zbinden & Lee 2005). Chen et al. (2009) found that off-farm income originating outside of the local area significantly increased the number of land plots re-enrolled into the Chinese SLCP PES programme. These results are replicated in a number of agri-conservation and FSM studies (Deressa et al. 2009; Dolisca et al. 2006; Knowler & Bradshaw 2007; Swinton & Quiroz 2003).

Moreover, Defrancesco et al. (2008) showed that a high dependency on farming activities for household income acted as a constraint to participation in AES schemes. A review of participation within conservation easement programmes also indicated a higher representation of part-time farmers over full-time ones (Kabii and Horwitz, 2006). In the marine setting, Gelcich et al. (2009) found occupational mobility to be a significant predictor of willingness to participate in the creation or administration of a locally co-managed Chilean MPA. In addition, empirical work by Cinner et al. (2009) suggests that fishers with greater access to alternative occupations would more readily stop fishing once stocks began to show a decline.

For the very poor an alternative income source can be the difference between a marginally viable livelihood and destitution. An income portfolio which best mitigates risk is one that has a low covariate risk between its components, e.g. the factors which create risk for one income source (e.g. climate) are not the same as the factors for another (e.g. urban job security) (Dercon 2002; Ellis 2000). However not all diversification strategies are created equal. In reality, poorer households are often marginalised from more favourable labour markets, which may require larger upfront capital, land or higher skill sets (Woldenhanna and Oskam, 2001). As a result, diversification by the poor still tends to leave them highly reliant on the exploitation of natural resources be it fishing, agriculture or the harvesting of 'wild' products; and these risks are not as uncorrelated as one would wish. In contrast the better off are more disposed to enter into less resource dependent activities such as trade, transport, shop keeping and small businesses (Ellis & Allison 2004).

It is predicted that fishers with a higher occupational diversity will be more likely to participate in the proposed marine PES for two reasons: one, fishers with alternative occupations will be better able to spread risk and so be less risk-adverse; and two, fishers with more than one livelihood are less invested into the sector and are able to redirect efforts and additional money to alternative occupations. It is further hypothesised that those fishers with investments into alternative activities not directly related to the environment – such as farming – would be even more likely to adopt the PES as the correlation between say business and fishing and farming and fishing would be less.

4.3.3.3 Social capital

A much less tangible concept, the role of social capital in household risk mitigation and consumption smoothing has also been well cited (Allison and Ellis, 2001; Coate and Ravallion, 1993; Dercon, 2002; Ellis, 2000; Fafchamps and Lund, 2003; Rosenzweig, 1988; Townsend, 1994). Broadly speaking social capital refers to the shared knowledge and understandings, social norms and bonds which facilitate collective action (Ostrom, 1999; Pretty, 2003; Woolcock, 2001).

The role of social capital, much like that of asset accumulation and income diversification, has been well documented within the participation literature. While its exact definition is subject to debate, broadly speaking social capital refers to the shared knowledge and understandings, social norms and networks which facilitate collective action (Ostrom, 1999; Woolcock, 2001).

The idea that social capital can influence an individual's adoption decision is situated within the theory of embeddedness. An alternative to the rational actor hypothesis, social embeddedness was first introduced by Polanyi in 1944. Revisited by Granovetter forty years later, Granovetter (1985) argues that "behaviour and institutions are so constrained by on going social relations that to construe them as independent is a grievous misunderstanding". Social embeddedness argues that market behaviour and decision-making are driven simultaneously by economic and non-economic motives and remain positioned within structures of social norms, social relationships and reciprocity networks (Breetz et al., 2005; Granovetter, 2005; Mariola, 2012). These social norms can play an important role in an individual's decision matrix. At its broadest level, these norms are defined as an understanding of how community members will behave under given circumstances (Chen et al., 2009). Moreover, reciprocity relationships have been suggested as playing a considerably more important role within poorer societal groups, where the possession of this social capital may enable access to goods and services often commoditised by higher

income groups (Fafchamps and Lund, 2003; Wakefield and Poland, 2005). Indeed, increased social capital has been shown to lead to greater risk sharing among villager members, acting as an informal safety net (Narayan and Pritchett, 1997).

Within the adoption literature, social capital and embeddedness has received considerable interest as it relates to the adoption of improved agricultural technology within low-income settings (e.g. Bandiera and Rasul, 2003; Bandiera and Rasul, 2006; Conley and Udry, 2010; Isham, 2002; Matuschke and Qaim, 2009). However, this literature investigates the role of social capital as it relates to the improved access to information and the reduced transaction costs associated with the dissemination of this information, as well as the perceived risk associated with new technologies previously tested by neighbours and counterparts. This may be less pertinent to the adoption of a marine PES scheme within this setting, which at the time of this study was in a hypothetical stage.

Within the PES literature, the influence of social capital on adoption has received much less attention. Recently, Ma et al. (2010) found spatial variation effects, possibly from interpersonal communications as well as other socio-economic factors. Although not at the individual level, Gong et al. (2010) found that lower levels of village social capital constrained participation in a PES forest project in Guangxi, China. In addition, Chen et al. (2009) found that social norms had a significant effect on an individual's intentions to re-enrol in PES projects at the village level, and that, aggregated, perceptions of neighbouring behaviour could substantially reduce PES programme costs. The intention to re-enrolment were shown to be highly affected by the re-enrolment decisions of neighbours and tended to conform to the majority: witnessing 10% of neighbours reconverting at least part of their SLCP programme plots back into agriculture was estimated to reduce a participant's intention to re-enrol by 6.4% on average. The effect of social norms was seen to be highest when payments were at an intermediate level; at higher and lower payment values it is anticipated that financial considerations override these effects. In a more recent empirical analysis of PES adoption, van der Horst (2011) cited the frequent occurrence of PES micro-clusters in environmentally sensitive areas as evidence for strong neighbourhood effects. In a fisheries context, Sesabo and Tol (2005) found a households' decision to participate in various income-generating activities was influenced by household structure and local institutions. In particular, the authors noted how households with higher social capital, in the form of access to social networks, showed increased participation in other livelihood occupations.

Perhaps most relevant to the question of social capital and adoption within our marine PES setting is the strand of literature focussing on how social capital enables individuals to take more 'risky' stances much as asset endowment does. And consequentially how this relates to willingness to adopt novel, and possibly risky, untested programmes. This strand has received considerably less interest within the conservation literature, and is almost lacking in how it can affect PES trading outcomes (Breetz et al., 2005; Mariola, 2012).

More generally, social capital as it relates to reducing risk has been cited to improve participation rates within AES, microfinance and CBM as well as investments into conservation agriculture practices (Nyangena, 2008; Pearlman, 2012; Polman and Slangen, 2008; Togba, 2012). Put simply, social capital captures the nature of social relationships and uses this information to explain observed behaviours and outcomes (Adger, 2003). It is however notoriously difficult to measure. In order to better understand (and indeed measure) social capital it is perhaps best thought of as five distinct but related dimensions (or proxies), as defined by the World Bank's Social Capital Implementation Framework. These are: 1) Trust; 2) Groups and networks; 3) Collective action; 4) Social inclusion and 5) Information and communication (The World Bank, undated).

4.3.3.3.1 Trust

At the individual level trust is the element which underlies the existence of social capital (Polman and Slangen, 2008). Trust is the mechanism which enables people to coordinate their actions for mutual benefit and as such overcome those market failures which arise through uncertainty (Ostrom, 1990). Quite simply the more people trust each other the more likely they are to contract with each other; trust lubricates cooperation serving to reduce transaction costs between individuals and buffers risk. Indeed a lack of trust will have negative consequences as all economic exchanges have an element of trust embedded within them (Mariola, 2012; Pretty, 2003; Sekhar, 2007). Trust is the most encompassing feature enabling collective action, and the other forms of social capital, for the greater part, contribute to successful collective action by enhancing trust between individuals. However, although trust among actors can often be explained as an outcome of other forms of social capital, it is also true that some aspects of trust are not reducible to these other forms (Ostrom and Ahn, 2001). For example, activity in voluntary organisations, although shown to increase trust between members, is only very weakly associated with generalised trust, where generalised trust is the measure of an individuals expectation of others

trustworthiness based upon more general information about social groups and situations. (Bjørnskov, 2006).

Trust in the implementing authority has been shown to be an important component in many conservation adoption decisions in both developed, developing and transitioning economies, for example JFM in India (Datta and Sarkar, 2010), AES in Europe (Polman and Slangen, 2008), as well as soil conservation initiatives in Eastern Europe (Prager et al., 2012) and Kenya (Nyangena, 2008). Similar reservations can be seen in PES initiatives. Recent reviews of farmer participation in water quality trading (WQT) schemes found trust to be an important determinant (Breetz et al., 2005; Mariola, 2012; Perrot-Maitre, 2006). Within these studies, a mistrust of regulators negatively influenced farmer's initial willingness to participate in the WQT scheme. In fact, the lack of trust in an environmental regulatory body has been shown to single-handedly stifle the adoption of conservation practices in a number of studies (Moore et al., 2008; Parker et al., 2009). The authors conclude that trust lubricates programme success through increasing the efficiency of outreach, improving the credibility of information and buffering the risks perceived by farmers.

It is not surprising that trust appears to play such a strong effect in the adoption of PES-like programmes. These financial incentives come with a number of perceived risks. In order to engage in a PES scheme, resource owners must often explicitly acknowledge that environmental damage is being done, and by them. This is certainly the case in the proposed marine PES herein: fishers must admit to those practices damaging the environment and concede fishing efforts as too high. In this way fishers can open themselves up to a number of risks: increased scrutiny; imposition of new regulations; possible loss of autonomy related to operations; increased government oversight and of course possible reneging on payment (Breetz et al., 2005; Langpap, 2004; Mariola, 2012). All this leads to a greater sense of uncertainty about future production values and profitability. Quite simply, the higher the trust the more a participant believes the scheme to function both in the long-term through trust in the information provided about the scheme, faith that undesirable effects will be mitigated and belief in the payment itself (Kabii and Horwitz, 2006).

Polman and Slangen (2008) analyse various types of trust as it relates to the design of various agri-environmental contracts within the European Union. Interestingly, general trust was seen not to have an impact on willingness to participate in any of the tested schemes. However, trust in government was seen a significant determinant in those cases where contracts imposed restrictions on intensive practices. In PES, more specifically, trust between buyer and seller has been argued as an essential

requisite (Corbera et al., 2007; Fisher et al., 2010; Vatn, 2010). Within a marine setting, a lack of trust in implementing bodies, and/or protected area managers has be demonstrated to lead to lower levels of compliance (Stern, 2008).

We hypothesise that trust will be an important motivation factor in marine PES adoption.

4.3.3.3.2 Groups and networks

Individuals come together to promote mutual interests and overcome mutual problems. Engagement in groups and networks allows those with common interests to benefit from coordination, conflict resolution, information sharing and building of common knowledge and trust (Pretty, 2003). Whereas group membership denotes an actor's participation, social networks relate to the structure of one's relationships and the types of connections involved.

Although social capital is created through interaction and more palpable at the group level through its network structures, it also has implications at the individual level (Uphoff, 2000; Woolcock, 2001). Social capital can be deconstructed into two separate but interrelated concepts: a) structural and b) cognitive (Uphoff, 2000). Structural social capital is associated with the various forms of social organisation, including the roles, rules, precedents and procedures as well as the assortment of network ties. At the individual level, cognitive social capital derives from the mental processes and resulting ideas relating to trust, reciprocity and learning. These are reinforced by culture, ideology and specifically the local norms, values, attitudes and beliefs, all of which contribute to cooperative behaviour and collective action (Uphoff, 2000). These two domains are intrinsically linked. Although true that networks with their roles, rules, precedents and procedures display a life of their own, ultimately they all come from cognitive processes, linked in practice though individual expectation. And it is this, in turn, which prescribes individual behaviour (Uphoff, 2000). Hence people's behaviour, experience and participation within groups and networks will have overall implications in their future choices and behaviour, as well as their knowledge of the expectations placed upon them and the expectations they place on others.

Group membership and presence within social networks has been shown as an important determinant in decisions to adopt novel agricultural practices (Bandiera and Rasul, 2006, 2003; Gabunda and Barker, 1994; Isham, 2002; Swinton and Quiroz, 2003). Swinton & Quiroz (2003) demonstrate a clear link between the number of household association memberships and the adoption of sustainable agricultural practices, or more specifically the area placed under fallow. However, much of the

adoption literature relates to the associated learning and reduced cost of information dissemination. Disentangling these effects from other group effects such as informal insurance networks is difficult.

Group memberships and network structure, quite simply, provide social ties. Social ties carry numerous benefits including the transfer of information and learning as well as reciprocal norms. These sharing arrangements can be viewed as implicit insurance structures (Nyangena, 2008). Food and/or financial assistance are extended through such networks. As discussed previously, these structures enable members to place higher values on long-term investments than non-members with similar flow and stock wealth. These networks have been shown to be important within rural and low-income societal groups (Fafchamps and Lund, 2003) as well as in fishing communities more generally (Bodin and Crona, 2008; Crona and Bodin, 2006; Isham, 2001).

4.3.3.3.3 Collective action

Collective action refers to instances where individuals work together for a shared good or benefit. When not imposed by an external force, collective action can be used as a proxy measure of underlying social capital (The World Bank, undated). For example, the degree to which one volunteers in a local group and/or attends community events is an indication of collective action. Many studies have shown collective action to be facilitated by social capital (Dasgupta and Serageldin, 2001; Krishna, 2003; Narayan and Pritchett, 1997; Ostrom, 1999, 1990; Putnam, 1995).

Whereas other forms of social capital mentioned refer to the structure of social relationships, collective action is perhaps better described as a flow associated with social capital; indeed it is a process which ultimately relates to its underlying social relationships. This dynamic nature of collective action, however, means that collective action is inherently hard to measure (Meinzen-Dick et al., 2004). A more comprehensive discussion on collective action can be found in Section 4.3.3.4.

4.3.3.3.4 Social inclusion

Social inclusion is the measure of an individual's inclusion in decision-making and collective action, as well as access to institutions (Oxoby, 2009; The World Bank, undated). One's degree of social inclusion will in turn influence the beliefs an actor might hold about access to these institutions and indeed any expected returns to social capital. Ultimately, the degree of one's social inclusion affects the incentives an individual faces when investing in social capital or in deciding how to behave within

society. Those with higher levels of 'social inclusion' might be expected to have a higher belief in the success of investments (Oxoby, 2009).

There is little empirical work explaining the role of social inclusion upon participation decisions but one might expect those with lower inclusion to be less able or likely to participate within conservation management activities. Those experiencing lower levels of inclusion are less likely to be involved within consultations and management decisions, nor are they likely to have access to many of the networks providing informal insurance.

4.3.3.3.5 Information and communication

The enhancement and maintenance of social capital ultimately relies on the ability of members to communicate with one another, as well as with others outside of these more closely tied networks. More specifically, information transfer through both network types has been shown to positively and significantly influence the adoption of technologies (Bandiera and Rasul, 2003; Bandiera and Rasul, 2006; Conley and Udry, 2010; Isham, 2002; Matuschke and Qaim, 2009); transaction costs and perceived risk are reduced. However, as previously mentioned this dimension may be less relevant for the adoption of the proposed marine PES given its extremely novel and presently hypothetical nature.

4.3.3.4 Determinants under a collective action scenario

Where PES schemes require collective action and indeed collective enforcement, participants must rely on not only their compliance but also that of others, as in the marine setting; other determinants come into play. While much of the previous literature review examines individual motivations, when looking at such a collective action mechanism additional determinants relating to interdependent decision-making will also be important.

A collective good is any good whose consumption is non-excludable. While individuals receive utility from the good, they are however also bearing the costs of its production. Under the theory of utility maximisation individuals will only participate in so much as their utility gains exceed the costs of participation (Fischer and Qaim, 2011).

There are a number of structural variables which predict the likelihood of collective action, these include: the number of participants involved; the heterogeneity of participants and information about past actions to name a few (Fisher et al., 2010). However, at the core of the ever-evolving theory of successful (or unsuccessful)

collective action are the degrees of trust between participants (Ostrom, 2010). And it is these structures and factors which enable people to trust one another and determine the level of trust (Nyangena, 2008; Ostrom, 2010).

Mutual trust has long been considered an important factor encouraging collective action (Ostrom, 1990; Putnam, 1995). Although the specific definition of trust in this context varies, scholars agree that the term encapsulates an additional and distinct causal force encouraging cooperation in isolation from institutional arrangements (Raymond, 2006). Herein we describe the term as "a willingness to take risks on the behaviour of others based on the belief that potential trustees will 'do what is right'" as defined by Hoffman (2002).

Research in various areas of collective action has shown the presence of, and development of social capital and trust to be important positive determinants in dealing with collective action problems (Emtage and Herbohn, 2012; Krishna and Uphoff, 1999; Nuggehalli and Prokopy, 2009; Nyangena, 2008; Ostrom and Ahn, 2001; Pretty and Ward, 2001; Pretty, 2003). Within the adoption literature, Nyangena (2008) presented results confirming trust as a risk pooling mechanism which significantly predicted adoption of soil and water conservation in rural Kenya. More specifically Emtage and Herbohn (2012) indicate that 'trust in others' is an important determinant of how farmers will respond to new natural resource management policies. However, more recently authors have questioned the importance of trust in collective action and point to the other necessary factor – institutional mechanisms which limit free riding and sanction non-compliers – as the facilitating factor (Cook et al., 2007; Raymond, 2006).

In rural coastal areas where enforcement and regulation are decentralised, and often lacking, the proposed marine PES will require a high level of compliance by local users. Under such a scheme sanctions can deter non-compliers; however, ultimately, the scheme will depend on the delivery of sufficient environmental services. Where resources are fluid, as they are in the marine environment, 'non-compliers' can do much harm. And where environmental service delivery is seriously injured it is likely that a programme will be discontinued; a statement to which effect is made within the questionnaire (as reported in Annex B1). Enrolment into such marine PES schemes may therefore be influenced by a fisher's belief in the compliance of others, or indeed by one's belief in the ability to free ride. For this reason it is anticipated that a fisher's degree of trust in his fellow fishers will have an influence additional to the individual characteristics previously mentioned. Trust within fishing families and small fishing communities has been shown to encourage individual fishers to observe

fishing rules and sustainable practices (Grafton, 2005). As such, one can hypothesis that this trust will also influence a fisher's likelihood to comply with and adopt any PES restrictions.

It is hypothesised that those fishers' with lower levels of trust in their counterparts will be less likely to sign up for any proposed marine PES scheme. By indicating low levels of trust in other fishers they will, in turn, anticipate low levels of compliance and a low return on any investment made into the common pool resources. On the flip side, fishers who believe their counterparts will also comply with PES restrictions will be more likely to sign up themselves.

4.4 Summary Hypotheses

Drawing on the adoption literature associated with environmental innovation and technology, conservation agriculture, agroforestry, microfinance, CBM and previous PES studies we identify a number of factors which may influence a fisher's decision to participate in a marine PES scheme. These are summarised in Table 4.1 along with the anticipated direction of the effect.

Table 4.1 summarised the hypotheses derived from the literature.

Attribute	Hypothesis	+/-
	Individual characteristics	
Age	Older fishers will be more resistant to change hence adoption of marine PES	_
Education	More educated fishers will be more willing to adopt marine PES	+
Household size	Assuming HHsize to be a proxy for a measure of dependents it is hypothesised that household size will reduce a fisher's likelihood to adopt novel schemes, hence the marine PES	_
Gender (where male=1)	The more restrictive nature of the scheme for women and the limited availability on their time outside of the home suggests women will be more resistant to the marine PES	+
Fishing income	Higher earning fishers will be more invested into the fishing industry hence more resistant to any restrictive marine PES	_
Fishing type (where legal=1)	Illegal fishers will be more resistant to the higher restrictions placed upon them	+
	Individual environmental beliefs and attitudes	
Positive attitudes to conservation	Those fishers who hold positive attitude to the environment and conservation will be more likely to adopt a marine PES	+
	Individual perceptions of risk and vulnerability	
Asset endowment	Those fishers with higher asset endowment will be more likely to adopt the novel marine PES scheme	+
Occupational diversity	Those fishers with great occupational diversity will be more likely to adopt the novel marine PES scheme	+
Social capital	Those fishers with higher levels of social capital such as trust/group membership and networks/social inclusion will more likely to adopt the marine PES scheme	+
	Collective Action	
Trust in fishers	Those fishers with higher levels of trust in other fishers will be more likely to adopt the marine PES scheme	+

Chapter 5

Site description

5.1 Overview

This chapter provides background to the study site. Data collection and methods are discussed in relevant chapters.

The chapter is organised into seven subsequent sections. Section 5.2 presents a summary of why one might consider a marine PES in the Mtwara coastal region. Section 5.3 provides an overview of the fisheries sector in Tanzania. Background information is subsequently provided on the Mtwara region in Section 5.4 and more specifically on the sampled villages which form the basis of the research presented in Chapters 6 and 7. Section 5.5 discusses Tanzania's enabling environment for marine PES. The proposed marine PES is laid out in Section 5.6 and relevant collaborations are described in Section 5.7.

5.2 PES in Mtwara, Tanzania

The Mtwara region in the United Republic of Tanzania offers an interesting case study for determining the suitability of a marine PES scheme within poor rural areas for the following reasons:

1. The coast and marine waters around Mtwara represent areas of high biological significant. A critical node for the accumulation and dispersal of marine organisms across East Africa the Mnazi Bay – Ruvuma Estuary represents an area of high biodiversity and replenishment value at both the national and international level.
2. High fishing pressure and destructive fishing practices significantly impact the health of the region's coral reef and marine environment. Destructive fishing practices – such as dynamite fishing – are used extensively, directly destroying the reefs and/or preventing recovery.
3. At present fishing along the coast is carried out under what is effectively an open-access regime. However, recent legislative structures in Tanzania promote local user rights and exclusion rules.
4. Small-scale fishery interventions can play a significant role in human and socio-economic development and represent an entry point for poverty reduction.

5. Mtwara is one Tanzania's largest urban areas and its population continues to expand with extensive migration into the area and a national annual growth rate of 2-6%. The region is considered among one of the country's poorest and least developed. Coastal communities have relatively poor access to public infrastructure such as water and sanitation, have little to no access to credit and tend to live in poor housing conditions.

5.3 The status of Tanzania's fishers

Tanzania remains one of the poorest countries in the world, ranking 152 out of 182 in the United Nations Development Programme's (UNDP) human development index (HDI) (UNDP 2011). As many as 34% of Tanzania's population are considered "basic needs poor[10]", 38% in rural areas. Official poverty levels have remained fairly constant over the last ten year but absolute numbers have increased (NBS 2007).

Tanzania's 30,000 km^2 coastal area currently supports a quarter of the of the country's 43 million strong population; a figure which is set to double by 2025 (World Bank 2011; Gustavson et al. 2009).

Marine resources make valuable nutritional and economic contributions to the communities living along the Tanzanian coastline and numerous islands. Marine capture fishing has long been regarded as one of the most important activities along the coast; the contribution these fisheries fluctuates between 2.1 – 5.0% of GDP for mainland Tanzania (Jiddawi and Öhman, 2002; Sesabo and Tol, 2005). Fish caught is primarily consumed on the domestic market and per capital consumption has been estimated at 25-30 kg $person^{-1}$ $year^{-1}$ (Jiddawi and Öhman, 2002).

The fisheries sector is almost entirely dominated by small-scale, low-income fishing households. Together these fishers account for 95% of Tanzania's total catch (Sesabo and Tol, 2005). In 2007, FAO reported approximately 150,000 registered artisanal fishers; of which over 20,000 were mainland coastal fishers; many other remain unregistered (FAO, n.d.; Jiddawi and Öhman, 2002).

Like many other coastal African countries characterised by open-access and traditional methods. Marine capture fishing is mostly restricted to inshore coastal waters: typically to within 4km from shore within waters along the country's narrow continental shelf. This is due to the limited range of traditional vessels and gear, and a lack of technical skills and capital to pursue gains in deeper waters (Masalu, 2000;

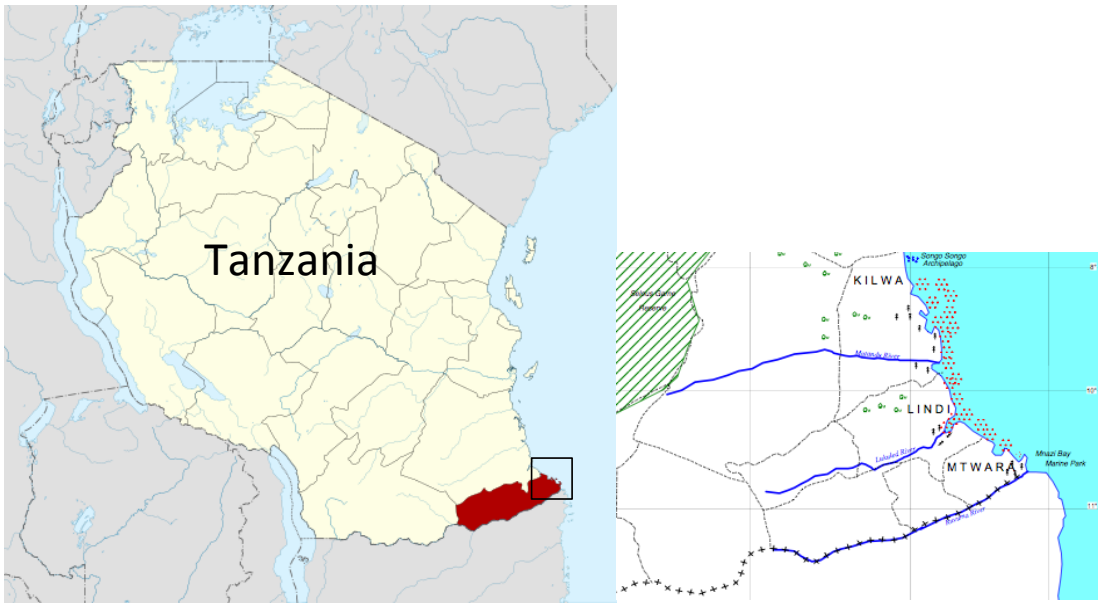
[10] The basic needs approach defines the absolute minimum resources required to satisfy long-term physical wellbeing. The poverty line is then defined as the income needed to meet these needs.

Sesabo and Tol, 2005). As a result, recent years have witnessed increasing pressures upon coastal resources and a poor performance in Tanzania's fisheries production (Sesabo and Tol, 2005). The country continues to experience crowding and overfishing of inshore waters (Jiddaw, 2001; Shao et al., 2003).

5.4 Mtwara, Tanzania

Located in the south of Tanzania and bordering Mozambique, Mtwara is the most southern of Tanzania's five administrative regions (Masalu, 2000). Mtwara is considered among one of the country's poorest and least developed regions, primarily due to lack infrastructure such as roads and energy. Thirty eight percent of the population live below the basic needs poverty line, with the coastal population considered amongst the poorest (Guerreiro et al., 2010; Malleret, 2004). Extending along 125 km of coastline are the region's two coastal districts: Mtwara Urban and Mtwara Rural. Together these two districts comprise around 26% of the Region's total population of 1.2 million: 92,602 and 204,770 respectively (Barr 2010; Guerreiro et al. 2010). The study area is highlighted in Figure 5.1.

Figure 5.1 Location of the Mtwara Region and study site within Tanzania



[1.] Mtwara Region shown as shaded area. Boxed area indicates coastal area and location of study sites. Adapted from http://en.wikipedia.org/wiki/Mtwara_Region (12 June 2011). [2.] Legend: + + + International border; ——— Rivers; ::: Coral reef. Taken from Shao et al., (2003)

5.4.1 Biological significance of Mtwara's marine environment

The Eastern African Marine Ecosystem (EAME) extends from South Africa to Somalia, crossing Mozambican, Tanzanian and Kenyan waters: a distance of 4,600 km

and encompasses an area of 540,900 km^2 once you include its 200 nautical mile economic exclusion zone (Guerreiro et al., 2010). Considered one of the top marine ecoregions for biodiversity on earth, the EAME supports some 21.5 million people who directly rely on the goods and services it provides (Guerreiro et al., 2011, 2010).

Mtwara's coastal waters are of high national and international importance. The area contains some of Tanzania's most significant biodiversity. It supports over 48 genera of scleractinian coral, 15 species of soft coral, 137 species of macro algae and 400 species of fish. It is also an important area for dugong, humpback whales, turtle, dolphins, birds and pelagic fish (WWF, 2004).

Moreover, this area is a major source of food and nutrients to adjacent waters. Its coral reef, which extends south from neighbouring region Lindi to the Mozambican border, connects with the Mozambican Quirimbas reef system. Together these reef systems are of critical importance as sources of marine larvae and spores which disperse out to northern and southern marine ecosystems; the Southern Equatorial Current diverges in this area creating an area of high replenishment capability (Shao et al., 2003; WWF, 2004).

5.4.2 Fishers of Mtwara

Households within Tanzania's coastline communities are generally large families with low per capita incomes and high illiteracy rates. Most coastal communities remain isolated due to poor infrastructure such as roads, electrical services and water supply (Gustavson et al., 2009). As few as 2% of rural housing have electricity, 6% have bank accounts, 25% have modern walling and as many as 45% must travel more than 1 km to access drinking water. Malaria affects 69% and 60% of children and adults respectively, and households are also face with other notable diseases such as HIV/AIDS, cholera and schistosomiasis (Gustavson et al., 2009; National Bureau of Statistics (NBS), 2002).

The livelihoods of coastal communities are highly dependent on natural resource extraction. Traditionally, livelihoods are based around subsistence and small-scale commercial activities such as artisanal fisheries, agriculture, mariculture, animal husbandry, salt and lime production, small-trade trade and crafts as well as mangrove and coastal thicket-related activities. Activities generally provide food, a source of shelter or provide limited income from local markets. A predominant feature of these coastal households is the need to be involved in several simultaneous livelihood activities to supplement incomes and maintain a consistent source of food (Gustavson et al., 2009). High and increasing poverty is prevalent amongst fishers:

average yearly income in most Tanzanian coastal villages does not exceed US$ 100 per person. Moreover, fish supplies per person are declining and excessive exploitation of the fishery continues (Cinner, 2010; Olale et al., 2010; Sesabo et al., 2006). Yet, the number of households participating in the sector continues to rise due to relatively high prices and a high demand for fish products (Bagachwa et al., 1994).

The coastal areas of Mtwara are no different, rural villages have no mains electrical supply, unreliable water supply and access to health, education and other basic services are only available in Mtwara town, a 20-40 km walk for some coastal villages. Communities largely rely on agriculture and artisanal fishing for their livelihoods, and annual per capita income remains below $100 a year (UNDP, 2002). Coastal villages all show a high dependence on marine resources (Malleret, 2004). This can be as high as 63 – 74% of households in some sea bordering villages with 54% of households directly depended on or were involved in fishing. Figures which are homogeneous and consistent with other studies across Tanzania and Kenya (Malleret and Simbua, 2004). Agriculture productivity is generally limited and on marginal lands within this coastal zone due to poor soils. As a result, households rely on a vast array of livelihoods to meet basic needs (Malleret, 2004). In these villages, fishing generally remains the main activity for male-headed households while female-headed households rely more so on agriculture. This said women do fish and fishing within the tidal zone can represent important sources of income (Gustavson et al., 2009; Malleret, 2004).

Fishing in the region is conducted entirely at the small-scale level utilising traditional methods. Vessels are mainly dugout canoes and some planked construction boats known locally as dhows; only 2-3% of boats are motorised (Shao et al., 2003). As such, fishing is generally carried out in shallow reef areas which are easily accessible from shore. Catch composition is multi-species and reef fish account for the majority of this catch. Most commonly species caught are demersal fish, followed by some large and small pelagics; crustacean, octopus and squid are also common. In some villages, where women fish with mosquito needs – locally known as tandilo – sardines can comprise up to 25% of local catch (Malleret, 2004; Shao et al., 2003). As in other regions of Tanzania, fishing is conducted using a wide variety of gear types (Malleret, 2004). These different gear types and the subsequent frequency of use are described in Table 5.1 and Figure 5.2 respectively.

While men engage in many different types of fishing, women generally only partake in ‘tandilo’ and gleaning activities. In the past ‘tandilo’ fishing involved catching small fish ‘dagaa’ from shore and timing depended on low and high tides. Tandilo

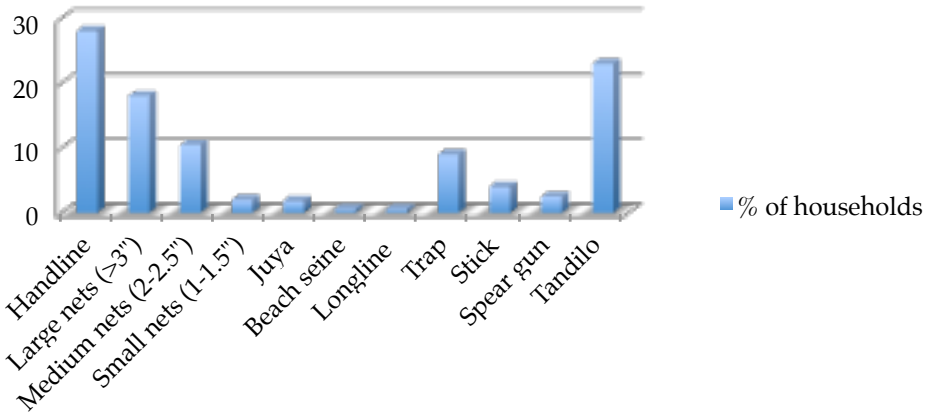
fishing is normally conducted within intertidal areas close to shore, using mosquito nests sewn together. Fishing for these women is part of a larger portfolio of activities and is generally conducted for a couple of hours each day for two weeks in each month. Women also involve themselves in gleaning molluscs from the shore as well as collecting sea cucumber and octopus during spring tides (Malleret, 2004).

Table 5.1 Gear types

Gear type	*Description*
Nets 4-7"	On average 50-150m in length and 20m in depth. Generally set overnight in deeper waters by 4-6 fishers. Target larger pelagic species such as shark
Nets 2.5-3"	From 50-100m long, usually set by boat and left overnight or day. Species targeted are mainly pelagics such as jack
Beach seine	Nets of very small mesh size (some 0.5"). Fishers based on beach spread seine over large area to encircle groups of fish. Non-discriminatory and mainly catches small fry and juveniles as well as larger demersal and pelagic fish
Tandilo	Mosquito nets used to fish close to shore. Nets are dragged along shore by 3-6 women. One of most widely used gear types within the park. Non-discriminatory, tandilo targets small fish including juveniles of larger species
'Juya'	Used as seines when groups of fish are spotted. May be in shallow or deep water. Fish are encircled in net, scraping bottom substrate when in shallower waters. Net is pulled and closed at bottom. Nets can be small (1-1.5") or larger (3")
Traps	These include traditional and fence traps. Fence traps can be left for days and are harvested at low tides. Traditional smaller traps are left overnight and mainly target demersal fish
Spear and sticks	These target lobster, octopus, sea cucumber and reef fish. Sticks and spears are used close to shore; spear guns can be used further out
Handlines	The most common fishing type among males, handlines are mainly used from boats and are often used alongside nets
Longlines	Longlines are fishing lines used from larger boats with multiple hooks. They target larger pelagics such as sharks. This gear is less common in the area

Adapted from Malleret (2004)

Figure 5.2 Gear use as % of fishers from household surveys: seafront villages



Source: Malleret (2004)

Mtwara's coastal communities are predominantly of Makonde Malaba ethnic origin, a coastal subgroup of the main Makonde tribe of southern Tanzania. The predominant faith is muslim (Malleret, 2004). The principle language within the region is Kiswahili, although Makonde is spoken extensively within areas along the coast. Polygamy is common within the area and most houses are male-headed. Culturally, women are not generally included within decision-making processes; men generally make most decisions and are involved in community decision-making forums. Compared with other women of Tanzania, women in this region are generally more confined to their houses. Female drop-outs from primary education are also commonplace (Shao et al., 2003).

Consistent with the rest of Tanzania, fisher numbers have risen within the region and productivity fallen (Harrison et al., 2010). In 1996, the number of registered fishers in the Mtwara coastal region was estimated to be 2050, approximately 10% of Tanzania's total registered artisanal fleet, in 2010 this figure was more than double at 5,600 (Dadi 2010). This number is anticipated to be much higher once non-registered male fishers are considered. Moreover, many women also engage in fishing activities and remain unregistered and outside of production figures.

A number of locally accepted fishing practices are now known to be very destructive and are illegal. These include the use of small-mesh seine nets to capture fish from the seabed and around reefs. Nets are weighted and dragged across reef flats or are pulled around coral structures damaging them and other reef life. This often includes the beating and smashing of corals to scare fish into nets (Tobey and Torell, 2006). Moreover, the small mesh results in the capture of many juveniles. Other methods include catching more sedentary species with sticks, which also results in the trampling of reef flats. Dynamite fishing also remains a problem in the area (Dadi 2010).

The female fishing method *'tandilo'* is one of the more destructive fishing types currently practiced within Mtwara's coastal waters. This fishing method, which uses small meshing to catch sardines locally known as '*dagaa*', results in both the trampling of reef beds and a high juvenile catch; the majority of the catch (61%) is below half the maximum adult size (Jiddawi and Öhman, 2002). However, it is not generally perceived as damaging by the local communities and can represent one of the only sources of income for these fishing women (Malleret, 2004).

In the last five years reported returns from fishing have significantly fallen. Declining stock and ascribed value (in Tanzanian Shillings: TSh) for fish catch in the Mtwara rural area are displayed in Table 5.2.

Table 5.2 Weight and value of fish catch landed by fishers in the Mtwara rural district for years 2004-2009

Year	*Amount of fish (Tons)*	*Value of fish (TSh)*
2004/05	120,240	83,825,290
2005/06	112,860	63,730,117
2006/07	94,321	65,755,586
2007/08	76,513	53,341,021
2008/09	61,245	56,639,934

Source: Mtwara Rural District office 2010, in: Harrison et al. 2010

5.4.3 Management of Mtwara's marine resources

Mtwara's shallow reefs (located within 1-10m from shore) are almost completely degraded having been heavily impacted by human impacts and fishers with a limited ability to access alternative areas (Shao et al., 2003).

In response to increasing environmental threats and high biological significance, the Tanzanian government gazetted Mnazi Bay-Ruvuma Estuary Marine Park (MBREMP) in 2000. The region's only marine park MBREMP is located to the west of Mtwara Region's main city Mtwara. The park covers 650 km^2, of which approximately 430 km^2 is sea, including mangrove forests, islands, seagrass and coral reef ecosystems; the remaining 220 km^2 is terrestrial (IUCN, 2005; Robinson et al., 2012). The 70 km^2 of mangrove forest found within the park accounts for almost 10% of Tanzania's mangrove forests (Wagner et al., 2004). Figure 5.3 presents the Marine Park's coastal border.

One of the Park's key objectives is to *"enable local and Government stakeholders to promote sustainable resource use and biodiversity within the park"* (Robinson et al., 2012). In 2004, eleven villages and three sub-villages were registered within the park[11]. The inclusion of the land and subsequent villages was to ensure that the local marine resource users were included within the management and planning process of the park, as required under Tanzanian law (Malleret, 2004).

Formed under the Marine Parks and Reserves Act (1994), MBREMP is under the control of the Marine Reserves Park Unit (MRPU). MBREMP has four active working departments which are headed by the chief warden, supported by senior wardens in

[11] MBREMP was originally established to be a zoned park, in practice to date no zones have been cordoned off as restricted use areas. In park fishers do however experience higher levels of monitoring and regulation of Tanzanian fishing laws.

each department. The departments include: administration, law enforcement, community conservation and research and monitoring (Robinson et al., 2012).

Figure 5.3 Map of local area indicating study villages, Marine Park and hypothetical closure sites

Marine park border shown outlined in thick yellow, hypothetical closures indicated by thinner white boxes. Taken and adapted from Google Earth (Version 6.0.1.2032) [Software}. Mtwara coastal view, TZ: Google Inc (2011). Available from: http://www.google.co.uk/intl/en_uk/earth/index.html

MBREMP is effectively a multi-purpose marine park, and continues to allow fishing within its borders. Regulations within the park are essentially the same as those outside, albeit enforced more frequently. These include: prohibition of certain destructive gears such as beach seine nets and dynamite; mangrove cutting for commercial sale; and the use of nets with meshing smaller than 3". Fishing within the park boundaries is restricted to artisanal fishers residing in those villages located within the park, however identifying fishers from within the park is difficult and proves hard to enforce (Robinson et al., 2012).

Six park rangers enforce park regulations and responsibilities include: day-to-day patrolling activities including park boundaries and scuba to ensure compliance; regular checks of fishing gears; as well as assisting the prosecution process. Non-compliant fishers risk confiscation of catch, fishing gears and boat (Robinson et al., 2012). Not surprisingly, these rangers are spread thin and probabilities of being caught remains low. Consequently, coastlines within the park, like those outside, continue to suffer from growing human pressures.

In order to reduce extractive pressure within the Park, park managers are promoting the development of alternative and supplementation income-generating activities alongside enforcement (Robinson et al., 2012). To date MBREMP has implemented a variety of interventions to improve livelihoods and reduce reliance on inshore resources. MBREMP's first initiative was a gear exchange in 2006. Villages were offered large mesh nets (5-6" mesh) in exchange for cheaper, illegal, small mesh nets. However, these nets proved inappropriate for traditional inshore fishing areas due to the lack of larger fish available; many fishers sold their new nets and reverted back to illegal alternatives. Subsequent gear exchanges took place in 2007 and 2008 with smaller 3" nets. However it still proved difficult to catch fish with these nets and little means of accessing deeper waters.

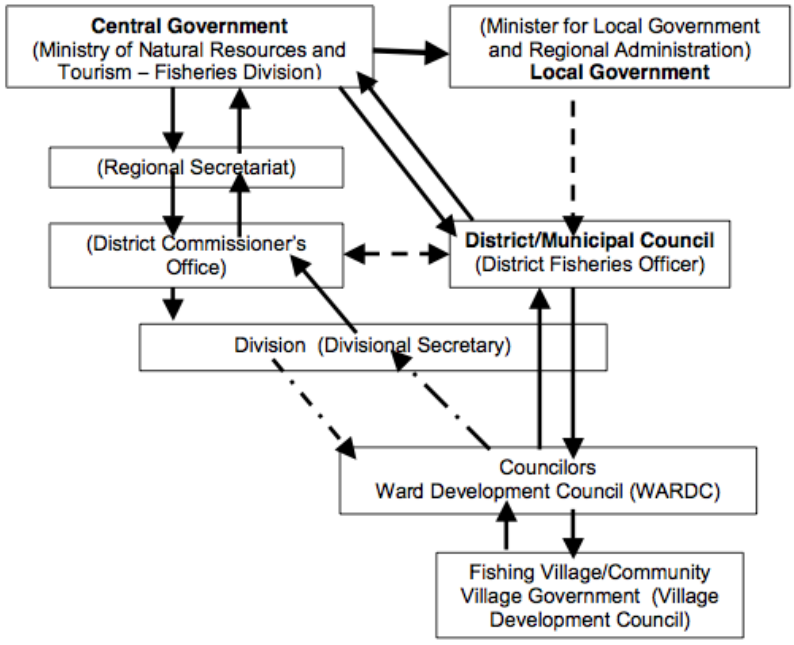
Additionally, MBREMP have looked towards implementing supplementary livelihood and fishing technology programmes which have been supported by a number of NGOs including World Wildlife Fund for Nature (WWF). These have included beekeeping, fish farming and livestock projects as well as a number of net exchanges which offered boat and engines alongside larger nets (due to cash constraints, however, this option was offered to only a few fishing cooperatives). Unfortunately, overall significant benefits have only been realised by a relatively small group of villagers (Robinson et al., 2012).

Outside of the marine park fishery enforcement falls under the mandate of the Fisheries Division of the Ministry of Natural Resources and Tourism. Under decentralisation, the central Government's role is to form policies and regulations which are implemented indirectly through the District/Municipal Councils. Supervisory roles go through the Regional Secretariat and the District Commissioner's Office. The Organisational environment is described below in Figure 5.4.

There are a number of policies and legislation relevant to the management of coastal and marine areas. These are described in greater detail in Annex B2. However, most important for current comanagement and potential marine PES are the recent changes in legislation to Tanzania's Fisheries Act (2003). These changes have seen the recent establishment of Beach Management Units (BMUs) at the fishing village level (Sobo, 2012). This change in legislation essentially allows communities control and management over their local inshore fishing grounds through the development of BMU. A BMU is made up of a BMU Assembly and BMU Committee. All persons engaged in fishing activities at the beach level must register within a BMU Assembly in order to legally access the fishery. Furthermore, BMU Committees comprise 9-15

members who are elected by the aforementioned assembly (Tanzanian Fisheries Division, 2005). These units are community fisheries organisations with legally empowered roles and responsibilities over local fishery legislation. In particular a BMU is *"able to set management rules locally and at lake wide level through by-laws and ordinance…. (and) allows control of access to fisheries resources by limiting number and types of fishing boats and gears in partnership with Government"* (Tanzanian Fisheries Division, 2005). Essentially BMUs are legally recognised comanagement instruments which devolve property rights over the coastal environment. The counterpart of BMUs within marine parks are Village Liaison Committees (VLCs). VLCs have fewer legal rights over resources as areas within marine parks are under the control of the MRPU. However, the MRPU's mandate is to establish and ensure sustainable conservation of areas of outstanding marine ecological importance, and to manage them *in partnership* with the coastal communities (Silva, 2006).

Figure 5.4 Organisation of Central and District Governments within the fishing community environment.



Source: Shao et al., (2003).

Continuous arrows indicate direct linkages; dotted arrows indicate indirect links with little implementation enforcement

However, within the region few BMU and VLCs exist; many villages have no BMU and those documented VLCs are not fully functional (Harrison et al., 2010). In reality, local fishers have, in effect, minimal influence on legislation and regulations concerning fishing or natural resource management more generally. Relationships are

strained between enforcing entities and local users generally. Collective efforts at the village-level remain low due to low levels of trust between administrators, enforcers and the fishers themselves (Bunce et al., 2010; Shao et al., 2003).

5.4.4 Village selection and characteristics

Villages were selected to give a representative sample of the area. A marine PES scheme will require participation by all marine resource users so it was important to collect data across a variety of village types. Villages were selected from both within and outside of the Park boundaries and with varying levels of dependence upon marine extraction, although overall all villages are considered to be on the high-end of dependence on marine extraction regionally.

Village selection was based on: the level of dependence on marine resources within the villages; high levels of extraction within proposed areas to be closed under the PES scheme; and ease of access[12]. Work by Samoilys (2010) and Yahya (2010)[13] identified areas of high biological significance to be closed under the proposed PES as well as those villages predisposed to access these areas. Six villages were chosen for sampling within Mtwara's coastal districts: three within the marine park (Mngoji, Mkubiru and Msimbati) and three outside (Mikindani, Naumbu and Pemba). Village location is displayed in Figure 5.3.

Table 5.3 presents village population counts and illustrates the variation in fisher numbers between the selected villages. Population sizes vary substantially from 912 to over 11,000, however a great part of this variation is due to the presence of two larger villages within the study. Fisher representation within villages also differs widely; as little as 7.3% of Mikindani's male population are fishers, in contrast to Mkubiru, Naumbu and Pemba where approximately all men fished. Similar patterns are seen within the female population, where again these three villages are seen to rely most heavily on the fisheries. It should also be noted that although the percentages do not look as large for Msimbati, this village actually contains the largest absolute male fisher population for any one village, and joint largest for women. These differences may in part be due to the relative isolation of Mkubiru, Msimbati, Naumbu and Pemba which are further away and with less developed connections to Mtwara town.

[12] Some areas were impassible during the wet season and village access was not possible.

[13] Data was collected during the spring of 2010. Reports were based on reef surveys, primary socioeconomic data and secondary data. Data was collected from within the Park and from areas north of the Park up to Sudi Bay, approximately 40km north of Mtwara town (south of Mtwara is Mozambique). Selection of villages was based on identification of those with high use within proposed closed areas. Proposed areas were areas of high biodiversity based on in-water transects.

Fisher surveys were conducted with 662 fishers, of which 35 were incomplete and omitted from analyses. A final useable sample size of 627 was obtained. Mean demographics of the sampled fishers are displayed in Table 5.4. Data presented below originates from collected fisher questionnaires implemented within the sampled villages. Methods are discussed in relevant subsequent chapters.

Average age of fishers was 35 and household size was 5.0, and comparable to the national average of 4.8 and 5.1 in rural areas.

Table 5.3 Summary of village characteristics

Village	*No. households*	*Total population*	*No of fishers (male)*	*No of fishers (female)*	*Fishers as % of total adult male pop*	*Fishers as % of total adult female pop*
Within Park						
Mkubiru	360	1540	400	300	100.0	77.9
Mngoji	570	1714	70	50	16.3	11.7
Msimbati	1120	10140	1521	300	60.0	12.3
Outside Park						
Mikindani	2777	11032	200	40	7.3	1.4
Naumbu	612	1758	600	150	100.0	34.1
Pemba	N/A	912	228	78	100.0	34.2

Source: Mtwara District Office (2010)

Table 5.4 Mean demographic characteristics fishers sampled

	In			*Out*			*All*	*In*	*Out*
	Mkub	*Mngj*	*Msim*	*Mkdn*	*Naum*	*Pemb*			
No.	162	81	106	47	124	107	627	363	264
Male (%)	41.4	43.2	70.8	72.3	48.4	46.7	51.1	48.8	54.5
Age	35.3	35.3	34.6	44.8	33.9	32.3	35.1	35.2	34.9
HH_size	4.8	5.6	5.0	5.3	5.2	4.7	5.0	5.0	4.9
Education (%)									
None	32.9	30.9	41.0	41.3	41.9	42.1	38.0	36.3	40.3
Primary	63.4	67.9	57.1	58.7	53.2	57.0	59.5	61.2	57.0
Secondary	2.5	0.0	1.9	0.0	4.0	0.9	1.9	1.7	2.2
Other	1.2	1.2	0.0	0.0	0.8	0.0	0.6	0.8	0.4

Where: In=villages located in park, Out=villages located outside of park; Mkub=Mkubiru, Mngi=Mngoji, Msim=Msimbati, Mkdn=Mikindani, Naum=Naumbu, Pemb=Pemba

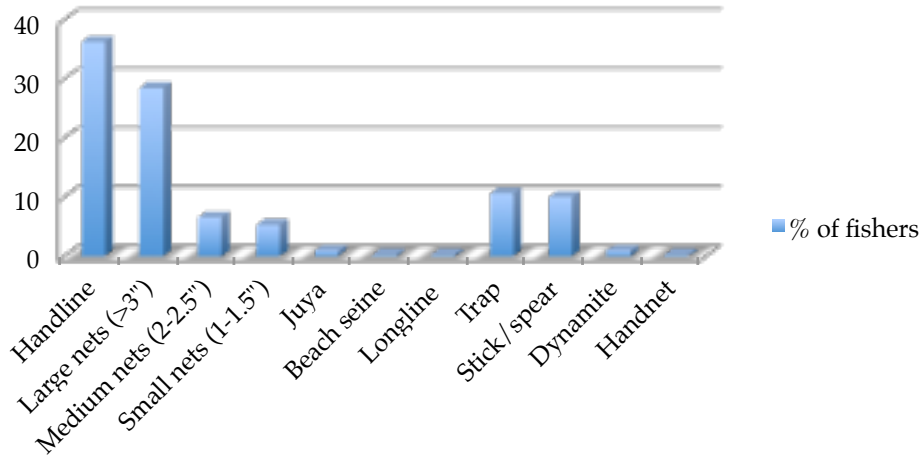
A number of fishing gears were sampled within the case site; the distribution is displayed in Table 5.5. The study aimed to survey men and women equally, as such

tandilo shows a higher percentage within the final sample than is representative of the area. Looking at male fishers independently a similar distribution of gear can be seen as found by Malleret (2004). Male gear use, as a % of male fishers sampled is displayed in Figure 5.5. The majority of male fishers used handlines or large nets, accounting for 65% of all male fishers interviewed.

Table 5.5 Fisher numbers broken down by gear

Gear type	*Number of fishers interviewed*	*% of total sample*	*% within Park*	*% outside Park*
Legal				
Handline	116	18.5	21.2	14.8
Large net (>3″)	91	14.5	11.0	19.3
Juya	3	0.5	0.6	0.4
Longline	1	0.2	0.3	0.0
Stick/spear	32	5.1	2.8	8.3
Trap	34	5.4	4.4	5.4
Handnet	1	0.2	0.3	0.0
Illegal				
Medium net (2-2.5″)	21	3.4	3.6	3.0
Small net (1-1.5″)	17	2.7	3.6	1.5
Beach Seine	1	0.2	0.0	0.2
Dynamite	3	0.5	1.1	0.0
Tandilo	306	48.9	51.2	45.5

Figure 5.5 Gear use as % of male fishers surveyed



Fifteen percent of fishers in the final sample earned 20,000 TSh or less a month from fishing (approximately US $13.8) and almost 8% earned on average only US$ 6.9 a month. The majority of fishers (61.1%) claimed to earn a monthly income between

20,000 and 40,000 TSh (US $13.8 – 27.6). These figures are consistent with previous socioeconomic studies in the area (Mallaret 2004) and national averages. National household fishing income figures are provided from comparison in Table 5.6.

Table 5.6 Monthly Fishing income in coastal regions of Tanzania

Monthly income from fishing	*% of sample*	
	Mtwara study site (individual earnings only)	*National figures (household)*
Less than 10,000	7.8	9.5
10,001 – 20,000	8.3	10.3
20,001 – 30,000	12.0	13.3
30,001 – 40,000	30.2	35.0
40,001 – 60,000	18.9	19.5
60,001 – 100,000	9.8	10.0
100,001 – 150,000	3.2	1.9
150,001 – 300,000	2.8	0.3
300,001 – 500,000	1.9	-
Above 500,000	2.0	-
Reluctant to divulge	3.2	0.3
Total (number of reponses)	100 (540)	100 (369)

(Adapted from Shao et al., 2003)

5.4.5 Poverty status of fishing households along Mtwara's coastline

The definition of poverty has evolved beyond a focus on only low income and consumption and is now known to be much more complex (Allison and Horemans, 2006). There are a number of varying definitions of poverty; poverty can mean different things to different people. Poverty is multi-dimensional and beyond merely having a low income. It includes facets such as: a lack of basic needs (e.g. access to food/shelter/health/sanitation); the lack of basic human rights; vulnerability and social exclusion; and feelings of powerlessness and humiliation. Fishing communities in low-income countries (as well as in those more developed) are often characterised by low levels of education, a lack of skills and physical assets, are highly exposed to accidents and illness, live in remote and isolated areas and are often without political voice (Allison and Horemans, 2006; Allison and Ellis, 2001; Townsley, 1998).

The Tanzanian National Household Budget Survey (NBS 2007) states that Tanzanian households are more likely to be poor if they are: large; have a large number of dependents; have a head who is economically inactive and depend on the sale of food and cash crops or earn a living from natural products rather that being part of the formal sector. Poverty is also strongly related to education; households where the head has above-primary education are on average five times less likely to be 'poor'

than those where the head has received no education. The ownership of a number of key household products has also been strongly correlated with reduced rates of poverty, these include modern roofing, radios and telephones, as well as access to improved an water supply, modern or improved toilet facilities and electricity.

As displayed in Table 5.7 education levels are low in the case study area, and consistent with regional estimates if a little lower than national figures. Overall, 38% of fishers surveyed had no education at all. NBS (2007) notes that nationally a quarter of the country's adult population still has no education, and further reports only 10% have secondary education or above. Regionally secondary school enrolment in Mtwara is approximately 2.5% (Harrison et al., 2010).

Table 5.7 presents further indicators from the sampled villages along side national averages. Average household size and percentage of population below 15 if fairly consistent with the national average.

Table 5.7 Summary poverty profile for fishing households

Indicator	*Total*	*Within Park*	*Outside Park*	*National Average (rural areas)*
Average HH size	5.0	5.0	4.9	4.8
Average % under 15s	45.3	43.4	48.0	43.9
% HH head with any education	69.8	72.4	66.2	76.4[14]
% of HHs with modern/non-earth walls	16.3	2.5	35.4	90.4 (15.6)
% of HHs with modern/non-thatch roof	11.9	9.8	14.6	55.5 (50.0)
% HH owning radio	55.5	52.0	60.2	66.0
% HH owning a cell phone	46.6	44.6	49.2	25.0
% HH with electricity	1.6	2.5	0.1	12.5 (2.7)
% HH with improved latrine	0.0	0.0	0.0	n/a

National averages: NBS (2007)

Key household materials appear lower than national averages. Only 16% of houses within the study site were made of 'modern' materials (e.g. materials more durable than earth), although this average is consistent with rural areas. Lack of modern roofing is more profound and lower even than rural averages. Differences are also seen for those villages found within the park and outside.

Household income was approximated using data collected on average income from fishing as well as other household activities such as cash crops, small business and livestock. Subsistence smallholder production was included into household budget

[14] Figure is for % adults overall with any level of education

using a multiplier of US $60 per hectare per annum for a mixture of selected root and cash crops[15]. Total household income was then adjusted as per number of household members. From these calculations it is estimated that approximately 42.7% of fishers within the surveyed areas are living below the basic needs poverty line. This percentage was higher for those communities living within the park (48.5%). More worrying still is that approximately 31.3% were found to be below the food poverty line, again with a higher incidence found within the park (37.2%). Results are shown in Table 5.8. This is much higher than the regional average of 17%. As can be seen from Figure 5.6 two thirds of households had earned the equivalent of 1,000 TSh or less per person per day, US $0.7. This percentage is again slightly higher for within park villages: 70.0% vs. 62.3%.

Table 5.8 Percentage of fisher households under poverty lines

	Total	*Within Park*	*Outside Park*
Below basic needs poverty line	42.7	48.5	34.9
Below food poverty line	31.3	37.2	23.3

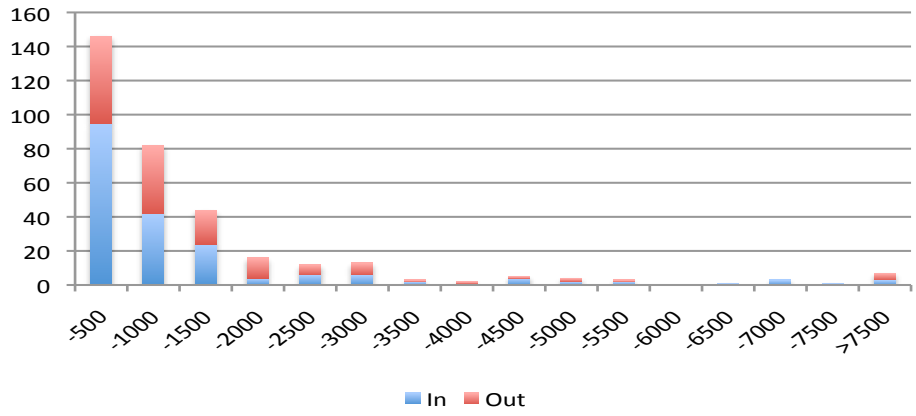
Table 5.9 Percentage of fisher households under poverty lines under a 25% error scenario

	Total		*Within Park*		*Outside Park*	
	-25%	*+25%*	*-25%*	*+25%*	*-25%*	*+25%*
Below basic needs poverty line	51.5	34.2	56.1	40.8	56.6	25.3
Below food poverty line	42.7	24.0	48.5	29.1	34.9	17.1

Given the fluctuating nature of income within these rural communities, as well as the complexities in collecting accurate income data, two alternative scenarios were also modelled. Assuming possible deviations of 25% from the calculated household figures gives results as presented in Table 5.9. Under the worse case scenario, over 50% of the fishers fall below the basic needs poverty line, and as much as 43% under the food poverty line. Even under the more optimistic scenario, 24% of fishers are still below the food poverty line, still much higher than the regional average.

Figure 5.6 Proxy income data per person in fisher households per day

[15] Value taken from Dobbin International, Spatial Development Planning Project: Perspectives on Regional Development. 2010.



Wealth ranking exercises were conducted with village-level focus groups which found similar patterns. In Mngoji, located within the marine park five wealth categories were defined: high income (5%); middle income & lower middle income (combined 25%); poor (50%); poorest (20%). In Naumbu, located outside of the park four wealth categories were identified: high income (1%); middle income (10%); poor (84%); poorest (5%). In both villages, the majority of fishers fell into the third wealth class or below: 52.6% in Mngoji, 90.1% in Naumbu. The poorest, bottom tier, members of the communities were comprised mainly of the elderly and disabled.

5.5 Why PES?

It is clear that the communities along Mtwara's coastline are causing the degradation of the marine resources, resources which have huge ecological benefit to a wider group of stakeholders. However, it is also apparent that fishers have few alternative options which would allow them to reduce fishing effort in the region. Ineffective boats, nets and training limit access to more productive fishing grounds and marginal soils prevent decent agricultural productivity. Providing compensation to fishers for reducing fishing effort within critical inshore fishing grounds can protect and restore currently degraded systems. This marine PES programme can be seen as a temporary solution which will benefit fishers in the short and long-term (as productivity will increase) but also regenerate an area of high national and international importance.

While it is not within the scope of this thesis to identify buyers, the sustainable use of Mtwara's marine resources and elimination of destructive fishing practices has the potential to benefit a number of parties. Within Mtwara's coastal waters destructive fishing and is the main cause of coral damage. Healthy coral reefs are important for not only sustaining local fisheries but also provide refugia for juveniles of pelagic species and are important in shoreline protection. The convergence of a number of currents in Mtwara's waters and its importance as an area for the dispersal of number

species further corroborates the broader regional value of this space. In addition a small but expanding tourism market is growing in and around the park boundaries and Tanzania is considered a worldwide tourism destination. If dive tourism is to grow in the area, the tourism values of the reefs, such as biodiversity and landscape beauty, will need preserving and protecting. It is also feasible that a market for 'blue carbon' exists in and around Mtwara. The park alone houses 10% of Tanzania's entire mangrove forests and contains rich areas of seagrass. Fishing practices are also known to cause trampling to local seagrass areas and mangrove clearing is a common problem (although not directly targeted within this proposed marine PES).

Tanzania has past experience with PES and PES-like instruments. Recently, Tanzania begun the piloting of terrestrial PES and has been selected as a pilot country under the UN-REDD pilot programme and the bilateral Norwegian government support for REDD reddiness work (Fisher, 2012; Fisher et al., 2012). As a result the Tanzanian government has been involved in a number of activities to prepare for REDD including capacity building and developing a national framework for these PES schemes (Fisher et al., 2012). Care International (CARE) and WWF Tanzania has also been implementing an Equitable Payments for Watershed Services (EPWS) project. Commencing in 2005, the EPWS programme in the Kibungo sub-catchment of the southern eastern Uluguru Mountains is now piloting a water PES which engages four upstream communities and two downstream buyers (Lopa et al., 2012). The success of this scheme however has not yet been fully analysed.

In addition, Tanzania has a well-developed history with Conditional Cash Transfers (CCTs). CCTs have a great deal in common with PES schemes and through their implementation have developed much of the institutional capacity needed for PES schemes more generally. CCTs are social 'security net' programmes which transfer cash to poor households on the condition that these households modify behaviour in some specified ways as to improve livelihoods. CCT programmes which impose a conditionality such as school attendance or implementation of specific health practices have been shown to be exceptionally effective in enhancing human capital within poorer societal circles (De Janvry et al., 2006).

In Tanzania, household-level conditional cash transfer programmes have been implemented under a community-based approach. Funded by the Japanese Social Development Fund, the Tanzania Community-Based Conditional Cash Transfer programme (CB-CCT) has been realised under the umbrella of the World Bank and Tanzanian Government's joint initiative: the Tanzanian Social Action Fund (TASAF) (Evans, 2008; Redko, 2013). Rolled out in 2010, the CB-CCT programme was designed

to enable poor and vulnerable communities fight poverty and promote economic growth as well as to prevent these groups falling further into poverty in the event of new shocks. Under the programme, 5000 households in 40 villages located across three Tanzanian districts were targeted (Kibaha, Bagamoyo and Chamwino). Conditions included primary school enrolment and 80% attendance for children aged 7-15 and regular check ups for children younger than seven (three times a year) and elderly (once a year). Participating families benefited US$ 3 a month per child and US$ 6 a month per elderly; a maximum of US$ 18 per month was set for each household. The community-based management of the fund had three levels: central; district; and community. The overall management and monitoring and support to subnational authorities was conducted at the Central Government level. Responsibility for the technical support, training and follow-up of implementation at the village level was held at the District level (Figure 5.4). However, selection of beneficiaries, day-to-day implementation, registration and verification of compliance as well as release of direct payments all took place at the community level (Evans et al., 2012). Recent results suggest the programme has had significant positive results including a reduction in sick days for treatment households and as well as an uptake of health insurance. Girls were also seen to be significantly more likely to complete primary school: 23 percentage points more likely than comparison group counterparts (Evans et al., 2012; Tanzania Social Action Fund, n.d.). In Tanzanian communities such CCTs have also proved effective in changing behaviour and promoting less harmful activities such as incentivising safe sex (de Walque et al., 2012).

Despite recent legislative changes in Tanzania which facilitate community co-management of CPRs, Mtwara's marine resources continue to effectively be accessed and managed as open-access. There are a number of reasons why collective action has not been successful (as described in Table 5.10). At present the conditions required for CPR management are not met within the coastal region of Mtwara and a number of changes must be realised before effective management will exist. Securing these changes will be vital for the successful functioning of PES scheme. These are laid out in Table 5.10

A marine PES programme can finance and incentive co-management of these CPRs. A PES can offer financing and promote institutional capacity for management, as well as provide appropriate incentives – enable communities some leeway to reduce extraction where needed in the first instance.

While all of the criteria laid out in Table 5.10 can be met, it is worth nothing that rebuilding trust and capacity will require dedicated time and effort by all parties. And, while previous lessons in CPR management do not necessarily preclude large areas from being successfully managed, the larger-scale and more abstract nature of the marine resource means that a good understanding of the resource and resource-users will be critical for PES design (Fisher et al., 2010; Ostrom, 1990)

5.6 The hypothetical marine PES programme

Two schemes were presented to the fishers. The first was a proposed PES scheme within the area which offered community fishers conditional cash payments as compensation for the closing of certain zoned fishing grounds and the discontinuation of all illegal fishing.

The scenario was presented as follows:

> *"CARE International, alongside WWF, as previously mentioned are interested in improving the marine environment in the area, as well as supporting the local livelihoods, particularly of the fishers who rely on these resources. In order to do so CARE International is considering a conditional cash transfer programme. A conditional cash transfer programme would mean that community fishers would be asked to not fish in those specific zoned areas as shown in the map as well as stop the use of all illegal fishing gear in all areas, but also compensated for their loss of earnings from these changes.*
>
> *Enumerators: Again show the respondent the map, identifying all the areas where fishing would not be permitted, also explain the scenario as described below.*
>
> *The scheme would originally run for 4 to 5 years. The payments would be conditional upon all fishers within this community, NOT fishing in these designated areas or using illegal gear. The payments would be made on a monthly basis, and all payments would be cancelled if fishing continued within the designated areas and illegal gears continued to be used. During this time additional investment will be made into the development of alternative occupations which will increase the availabilities of alternative activities in the area. Monitoring would be a combined effort between local communities, who would all lose out if the rules were broken and the Marine Park authority inside park/BMUs officials outside park. In this time it is expected that fish stocks will have suitable recovered and fishing profits increased, as well as management practices improved allowing the long-term sustainability and profitability of the fishing.*
>
> *For example far away in the Pacific Ocean, closing areas to fishing has increased local fishers catch, both in size and amount. The improvements took a few years to be seen, as fish within these areas require time to mature and grow, however after this they often leave the protected area. This is why CARE International is looking to support the programme with conditional payments over the time it is required for the stocks to recover. However, now these small fishing communities in the Pacific*

which have seen larger catches, more fish and higher income than neighbouring communities without such protected areas."

The scenario was presented as a joint intervention to be carried out via the International NGO CARE and WWF. The scenario was presented this way as at the time of the surveys CARE and WWF were in the process of conducting a feasibility study into such a marine PES and therefore would have been the likely facilitators if such a marine PES were to go ahead.

The core areas were chosen as these had been previously identified via in-water surveys as areas of high biodiversity and ecological significance, for example areas of high replenishment value. The hypothetical closures in question are displayed in Figure 5.3. The core zones to be closed were selected for the hypothetical scenario based upon the likelihood of which zones would likely be closed if the proposed marine PES were to go ahead. The core zones selected within the marine park were based on previous work by IUCN within the area. Zones were mapped through a participatory zoning workshop based on scientific data and representation of key ecological sites as well as input from communities (IUCN 2005). Outside of the park, core zones were selected for their similarity to core zones within the park, based on ecology and use from communities (Yahya 2010). The core zones to be closed represent a number of ecological sites including fringe reefs and intertidal zones. The zones within the park and outside of the park are for the most part used by the surrounding villages equally, segregated for the large part by those villages within the park utilising core zones within the park and those outside of the park utilising those located outside of park boundaries. This said a little overlap has been recorded but is more common for those villages outside of the park but closer to park boundaries. The most common differentiation would be the use of intertidal areas, which is most commonly used by women tandilo fishing. This said, women are known to also visit the fringing reefs at low tide, accessing such areas via boat.

In addition, the scenario calls for all illegal fishing to cease. The majority of illegal gears in use are netting with mesh sizes below the legal minimum of 3". The smaller meshed nets are responsible for catching of undersized fish and/or juvenile species. In addition tandilo and beach seining are responsible for the destruction of coral, mangrove and sea grass areas.

The proposed marine PES is also likely to deliver a various types and levels of environmental services per fisher depending on the fisher's own methods. Yet as previously discussed if all fishers are not considered within the marine PES, it is possible that non-participating fishers will absorb any gains made. Moreover, the

hypothetical PES has an element of poverty reduction within its design; that is the targeting of poor coastal communities to provide marine environmental services. As such, within this proposed marine PES all fishers are targeted due to: the mobile nature of the resources; a history of elite capture in the area; as well as to inspire compliance and reduce perverse incentives. The hypothetical PES also has an element of poverty reduction within its design; that is the targeting of poor coastal communities to provide marine environmental services. More targeted PES schemes are indeed possible but are not investigated here as this trade-off between efficacy and poverty alleviation are considered likely for most marine PES.

The payments were described be conditional upon all fishers within this community, not fishing in these designated areas or using illegal gear. The payments would be made on a monthly basis, and all payments would be cancelled if fishing continued within the designated areas and illegal gears continued to be used. During this time additional investment will be made into the development of alternative occupations which will increase the availabilities of alternative activities in the area. The presentation of payments being conditional on all fisher compliance was considered appropriate given the discussion in Chapter 3 and 4 of this thesis. Under such fluid marine resources, all (or at least a high proportion of) fishers will need to comply in order to deliver the necessary environmental services. Moreover, it is likely that a reduction in the number of fishers – in the longer term – will also be required hence an investment by CARE into the development of additional occupations in the area.

Payments would be made to the individual but through some village association such as the BMU or VLC. This was seen as the most reasonable distribution mechanism given the large possible number of fishers and the role that these BMU and/or VLC would have in monitoring and enforcement. Given the current legislation in place, as discussed in Section 5.4.3 closures and exclusion of others can be put in place through these BMUs and Marine Park Authority (in collaboration with the VLCs within the park). Enforcement within this scenario only speaks to the removal of payments and cessation of PES scheme and does not include sanctions such as confiscation of gear and/or boats.

The scheme was designed to originally run for 4 to 5 years as this was considered a reasonable timeframe to enable sufficient recovery of the marine resources and implementation of long-term sustainable practices and the expected time in which fishers would be able to recover sufficient earnings to no longer require a cover payment for loss of earnings. As described in Chapter 3 of this thesis marine PES can be used as a transitory instrument to enable the immediate losses of fishers when

restrictions to improve marine environmental services require short-term losses to said fishermen. However, these can also run along sustainable management plans which can protect the marine environmental services and provide sufficient catch once fisheries have recovered to a sustainable level.

It is anticipated that males and females could respond differently to the proposed marine PES. For example, at first glance the marine PES scenario presents very different outcomes for male and female fishers. For example, marine PES will call for a complete ban on tandilo fishing for women. However, tandilo represents only one type of fishing in which women are involved, women also commonly practice gleaning and octopus fishing which under the proposed marine PES scheme will still be viable. Moreover, the implications for within the male fisher group shows a similar pattern, for example some men will be required to stop fishing all together using their current practice, and although they can still fish using legal methods, transferring across to these fisheries is not generally as easy as one might imagine and barriers exist. Alternative activities to be invested to might include fish farming, livestock rearing and small business development, activities which have been previously tested within the area and represent good gender balances. Moreover, fishers expect to receive compensation for their own individual opportunity costs under this scenario. For this reason the scenarios presented to the male and female fishers are not as dissimilar as one might originally think.

The second distinct CE analysis – and the basis for the results presented within Chapter 7 of this thesis – presented a much less prescribed scenario. The notion behind this chapter was to determine those attribute of PES design which would influence participation and so variations across the number of restrictions within a marine PES scheme were presented. More information the various scenarios presented are available within the relevant chapter, Chapter 7.

5.7 Collaborators

This PhD was in part conducted in collaboration with CARE and WWF Tanzania. CARE is an international aid organisation focusing on tackling the underlying causes of poverty and targeting vulnerable groups with a particular emphasis on women. CARE acknowledges the need to address environmental problems in order to improve the lives of the rural poor. In 2011 the organisation worked in 84 countries, supported over 1000 poverty related programmes reaching more than 122 million.

CARE is a confederation composed of twelve national Members, each an autonomous non-governmental organisation in its own right. CARE Secretariat which is based in

Geneva, Switzerland coordinates and supports the work of the CARE national Members and Country Offices worldwide[16]. Work was conducted alongside staff based in the East African Regional Management Unit.

WWF is one of the world's leading international non-governmental conservation organisations. WWF focuses on issues relating to the conservation, research and the restoration of the environment. WWF works in 100 countries and supports over 1,300 conservation and environmental projects globally[17]. WWF has an local office base in Mtwara.

In 2010, CARE, in collaboration with WWF, commissioned a study to investigate the possibility of introducing a marine PES scheme to reduce community dependence on and exploitation of coastal fisheries along the coast in the Mtwara Region, both inside and outside of the marine park boundaries.

Financial assistance was provided by CARE. Logistical support was provided by WWF Mtwara.

Additional data collected alongside the results presented within this thesis were used to write a feasibility report for marine PES in the region. Results of the report were presented to CARE, WWF, Tanzanian fisheries officers and Marine Park officials at a National workshop in Dar es Salaam, Tanzania. A final report was submitted to the East African Regional Management Unit, CARE.

[16] For more information on CARE please see: http://www.care-international.org
[17] For more information on WWF please see: http://worldwildlife.org

Table 5.10 Ostrom's design principles[1] for successful CPR management and level conditions satisfied within Mtwara region

Ostrom Condition	Condition currently satisfied	Possible resolution	Steps required to establish necessary condition in Mtwara area
1. Clearly defined boundaries (effective exclusion of external unentitled parties).	No	Yes	Under present regime resources are effectively open access to local and nearby communities both inside and out of park boundaries, fishing by more distant migrant fishers also occurs. Within MBREMP the Marine Park authority can exclude fishers from specific areas, and enforce closures and community access rights. In fact the mandate states that non-park villages are not permitted to fish within park boundaries. This will require joint monitoring and improved relations between communities and park authorities. Requires conflict resolution, support and communication between communities and Park. Under current Tanzanian laws, BMUs have right to define community areas and enforce restrictions to outsiders (as well as community members). Exclusion therefore possible with capacity building. Outside of the Park a greater sense of ownership over neighbouring reefs is required. Communities have rights to exclude users under the BMU. At present no laws are in place to allow expulsion of non-village fishers and there is no BMU in place. Will require legal recognition of tenure under BMU legislation however to date no BMU is in place and management is realised or practiced.
2. Rules regarding the appropriation and provision of common resources are adapted to local conditions.	No	Yes	Within initial park proposal community consultations took place in order to define no-go areas as well as potential park programmes including benefits to be delivered at local level. Park has not delivered on its promises, at least not to the scale expected and additional restrictions have compromised ability of fishers to continue fishing as the once did. This has led to increased conflict and low levels of trust within the park and increased suspicion of governing bodies outside. Outside of the park no BMU agreements have been established with the District Fisheries Office. New programmes should be co-designed and co-management. BMUs and VCLs can enable consultations and joint-decision making between village representative and government officials. Co-design is possible but will require time and effort to rebuild previous relationships.

[1] Ostrom, Elinor (1990). *Governing the Commons: The Evolution of Institutions for Collective Action*. Cambridge University Press.

Ostrom Condition	Condition currently satisfied	Possible resolution	Steps required to establish necessary condition in Mtwara area
3. Collective-choice arrangements allow most resource appropriators to participate in the decision-making process.	No	Yes	Within the MBREMP communities VLCs mediate between Marine Park and communities. Villages outside of the park rely on BMUs to communicate with the Fisheries Division. Within the park VLCs have broken down and many members of the communities are unaware of their members or exist. Outside of the park the premise of BMUs are known about but few exist furthermore conflict due to confiscation of gear exacerbate relations between fishers and fisheries officials. Reestablishment of trust and conflict resolution is required. WWF is working with fishers within the park to create fishers forums and give them a stronger voice in decision-making. Resolutions possible are with capacity building. Further capacity building needed to better involve fishers and women in decision-making process
4. Effective monitoring by monitors who are part of or accountable to the appropriators.	No	Yes	Resources are limited and community involvement and compliance low. Within legislation BMUs are mandated with assisting the Fisheries division with data collection and monitoring. However as previously mentioned few effective BMUs exist. Reestablishment of VLCs within parks and BMUs outside as well as greater involvement of individual fishers in monitoring required. Creation of fisher forums inside and outside of park.
5. There is a scale of graduated sanctions for resource appropriators who violate community rules.	No/Some	Yes	Within MBREMP no community rules exist regarding marine resource use. Communities express wish for greater enforcement on illegal fishers but unable to act. Outside of park graded sanctions on illegal fishing exist, where third offence results in confiscation of gear. Actual practice of this debatable. Capacity building and conflict resolution are required. Strengthening of fishers' position to monitor and report (anonymously or not), as well as enforce gear restrictions is required. Again conflict resolution and communication strategies between Marine Park and communities required.
6. Mechanisms of conflict resolution are cheap and of easy access.	Unknown	Yes	At present no conflict mechanisms are in place. Fishers report conflicts mainly with illegal fishers and other village members but are unable to resolve. Again strengthening of relations between villagers and enforcement officers as well as improving communications between villages is required. Fisher forums, which outreach to other villages is recommended. Capacity building and community acceptance will reduce costs.
7. The self-determination of the community is recognised by higher-level authorities.	No	Yes	Limited communication exists. Communication structures between village level entities and government departments is practically non-existent. Again communication mechanisms and conflict resolution required.

Ostrom Condition	Condition currently satisfied	Possible resolution	Steps required to establish necessary condition in Mtwara area
8. In the case of larger common-pool resources: organization in the form of multiple layers of nested enterprises, with small local CPRs at the base level.	Some	Yes	At base level fishing cooperatives currently exist both within and outside of MBREMP. Fisher forums run one level higher. However, connection of these with higher authorities is low. Capacity building required.

Chapter 6

Determinants of fishers' willingness to adopt a marine Payments for Environmental Service scheme

6.1 Overview

We analyse the role of social capital in the willingness to adopt marine PES broken down by gender. We find that the various attributes of social capital influence the sexes' willingness to participate differently. More specifically, women showed a more profound reluctance to upset pre-existing social networks. The variable 'presence within a reciprocal fishing dependency network' was again seen to be significant negative determinant for women. The only social capital variable which was seen to significantly influenced men was trust; men with more pronounced trust overall were more likely to participate in the proposed marine PES.

The paper proceeds as follows. Section 6.2 provides a brief introduction to the topic. Section 6.3 presents an overview of fisher characteristics, poverty and the implications of these for marine PES schemes. Included is a discussion on the current treatment of women within fishery policy development and resource conservation interventions. Section 6.4 introduces the case study and Section 6.5 the methods. Descriptive results are displayed in Section 6.6. Section 6.7 lays out the econometric strategy and findings are presented in Section 6.8. Section 6.9 provides a discussion of the results and final conclusions are drawn in Section 6.10.

6.2 Introduction

The fisheries and aquaculture sector is estimated to provide direct employment and revenue to as many as 200 million people globally (FAO, 2012). In the developing world, small scale artisanal fisheries are important sources of employment, financial revenue for many, including some of the poorest and most vulnerable sectors of society (Béné, 2009; Johnson et al., 2013; Olale et al., 2010; Walmsey et al., 2006). Yet these small-scale fisheries are one of the major activities affecting coastal ecosystem health. Intense poverty and a lack of viable alternatives drive ever-increasing pressure on the artisanal fishing sector, which, unfortunately show few signs of abating.

Growing coastal populations, intense poverty, tenacious overfishing and the increased use of destructive fishing practices degrade coastal ecosystems and weaken their

capacity to provide beneficial environmental services both now and in the future both to these fishers and others (Berkes et al., 2001; Defeo and Castilla, 2005; Halpern et al., 2012). Small-scale fisheries are a key factor affecting coastal and coral reef health and most unregulated coastal artisanal fisheries are now considered overfished or collapsed (Defeo and Castilla, 2005; Hawkins and Roberts, 2004).

In as little as two decades, due to increasing recruitment and the widespread adoption of motorisation, small-scale fisheries have grown significantly, exerting increasing pressure on a finite resource base and the ecosystem. This has not, however, led to labour displacement; quite the contrary, these technical developments appear to have increased fishing days, and promoted greater employment within the sector (Mathew, 2001). Since 1970 it is estimated that the sector has witnessed a doubling in the number of fishers (FAO, 2005). Between 2005 and 2010, employment within the fisheries sector continued to grow at a rate faster than traditional agriculture and indeed that of the world's population, 2.1% annually vs. 0.5% and 1.2% respectively (FAO, 2012). This overfishing has resulted in a growing use of destructive fishing, the removal of large ecologically important species and the consecutive targeting of smaller species, and ultimately affects the resilience and services of this important ecosystem (Edinger et al., 1998; Pauly et al., 2002; Pikitch et al., 2004).

Unfortunately, the economic values provided by many of these environmental assets and services are not captured by these artisanal fishers; consequently these individuals lack many of the incentives to sustainably manage these marine environments for their production (Engel et al., 2008; Nelson et al., 2010). In order, to correct this market failure, recent conservation initiatives have looked towards new instruments: market-based instruments. These tools require that those benefiting from the environmental services compensate those who provide them, and are thus hailed with the ability to address those market failures which result in the under provision of vital environmental services (Engel et al., 2008; Wunder, 2005). And as a result, these initiatives continue to attract growing attention in policy, non profit, private and financial arenas (Engel et al., 2008; Mandel et al., 2009).

It is highly probable given the collective action nature of marine resources that marine PES schemes will rely on contracts made with 'fishing communities', whether that be fishing fleets, a specific fishery, or an artisanal community. Under democratic rules, participation will rely on the support of a majority of fishers. The premise that PES schemes are voluntary implies that potential actors will simply refuse to participate, or withdraw, if benefits are insufficient (Pagiola et al., 2008; Wunder and Albán, 2008). However, this decision to adopt a marine PES scheme will in fact be influenced by

several considerations, and not merely by the financial rewards offered (Allison and Ellis, 2001; Sesabo and Tol, 2005).

In order for PES programmes to benefit the coastal and marine environment in question, these fishing communities, and the fishers within them to which we refer, must not only be eligible and able to participate – whereby they possess all the requirements which enable them to enrol – they must also be willing to participate (Pagiola et al., 2005; Tschakert, 2007). Moreover, marine PES, while voluntary at a community level, may be obligatory at an individual level; therefore there will be a subsection of non-willing participants who will ultimately face a welfare loss.

In order to design more successful development-conservation programmes, there is a need to better understand the factors motivating adoption behaviour; chiefly how these relate to the decision to adopt new livelihood schemes, such as marine PES and PES more generally, as well as identifying those who consider participation a welfare loss. However to date, few attempts have been made to understand the key attributes which drive participation within PES (Kosoy et al., 2008), and none within the marine context.

This chapter investigates the association between fisher attributes, including perceptions of risk and vulnerability (e.g. income diversification and social capital), and stated willingness to participate in a hypothetical marine PES scheme, based in a low-income coastal community in southern Tanzania. The marine PES scheme is presented as a voluntary scheme whereby payments are contingent on compliance by all fishers currently extracting the resource. As such, the hypothetical scheme further presents some interesting insights for marine PES in promoting collective action as well constraints at the individual level. The paper serves to highlight the importance of non-monetary attributes that may play a role in the decision to adopt any novel PES scheme, and which without a proper understanding of PES schemes, marine or not, run the risk of further marginalising vulnerable and key target groups.

The paper is set out as follows. In the following section, 6.3, we provide further detail on the key characteristics of adoption as discussed in Chapter 5, including gender roles and household coping mechanisms as they relate more specifically to fishing communities. Section 6.4 presents the case study and Section 6.5, the survey design and implementation. Descriptive results are displayed in Section 6.6. Section 6.7 sets out the econometric strategy and Section 6.8 the subsequent findings. A discussion of these results follows in Section 6.9 and conclusions are presented in Section 6.10.

6.3 Fisher characteristics, poverty and implications for marine PES

6.3.1 Previous determinants of conservation success in fishing communities

Over the years much has been reported within the marine resource management literature relating to success of marine policy interactions and the promotion of support for these tools, i.e. with respect to MPAs. Socioeconomic and demographic characteristics of fishers have been shown to be important determinants in the acceptance and positive perception of marine resource management (Baticados 2004; Cinner & Pollnac 2004; McClanahan et al. 2009; Pomeroy et al. 2005; Sesabo et al. 2006). More positive views on restrictions have frequently been associated with age, expenditure and education, as well as more complex factors such as past experiences, existing values, attitudes and social norms (Aldon et al. 2011; Gelcich et al. 2005; McClanahan et al. 2009).

With respect to marine PES adoption, the literature remains silent. This being said, Chapter 4 identifies those characteristics important in adoption decisions more generally as well as, where possible, how it relates to fisher participation and exit decisions within marine management plans. In addition to the more general determinants examined in Chapter 4, there are a number of additional key features associated with fishers and their communities which may be important in the design any marine PES, and may have serious inferences in a fisher's decision to adoption. These are highlighted below.

6.3.2 Mechanisms to cope with insecure supply and high variability in catch

Small scale artisanal fisheries are defined by high levels of variability, some predictable and seasonal, others not. Few land-based occupations risk the loss of all productive capital, as well as participants' lives, every time they go to work (Béné et al., 2010). In general, these groups experience high exposure to natural, physical, health-related, climate induced and economic shocks and disasters (Mills et al. 2011; Béné 2009). Moreover, mobile prey and weather conditions result in high day-to-day variation of scheduling, catch and income (Pollnac, 1991), and fishers frequently experience economic reversals (FAO, 2001).

In response to these uncertainties of supply, fishing communities have developed numerous adaptations to smooth consumption and manage risk. Livelihood strategies such as income diversification and embedding oneself within social institutions based on trust, reciprocity and agreed norms play an important role in traditional artisanal fishing communities (Coate and Ravallion, 1993; McGoodwin, 2001). Fishing households often diversify into other economic sectors in order to spread risk and

improve one's ability to withstand shocks and resource variability (Allison and Ellis, 2001). Collaboration has been embedded in numerous forms of local associations and groups which allow actors to learn and diversify into new areas with greater security (Pretty, 2008). Collective care mechanisms have emerged which anticipate and minimise risks, such as kin, neighbourhood and community income redistribution mechanisms (Kurien and Paul, 2001; McGoodwin, 2001). In addition; fish buyers, familiar with the environmental constraints, provide loans with flexible repayment rates (Pollnac, 1991), and similarly, shopkeepers may extend credit in times of low catch.

The implications of these risk-mitigating strategies are examined further in Section 4.3.3 of this thesis.

6.3.3 Gender disparity in fisheries

Another attribute important in fishing communities, but largely overlooked, is gender. Gender is defined as 'socially determined ideas and practices of what it is to be female of male' (Baden and Reeves, 2000). Within fishing communities larger gender disparities exist (Bennett, 2005). Differences are seen in fishing strategies, motives and capabilities, as well as the ability of women to participate and benefit from new fishery development interventions (Allison and Ellis, 2001; De Silva, 2011; The World Bank et al., 2009). In reality, policy changes promoting commercialisation have further displaced women from established fishery roles (Porter and Mbezi, 2010).

For a long time the fishery sector has been perceived as a 'male only' domain and continues to be seen as so (Sze Choo et al., 2008). However, as can be seen this is a gross simplification of a very complex environment.

Underrepresented and largely ignored within the fisheries literature, women comprise approximately half of the fisheries and aquaculture workforce (FAO and World Fish Center, 2008; Johnson et al., 2013). The Big Numbers Project[19] revealed that for nine explored case study countries, 47% of the entire sector's labour force was women; this figure was as high as 73% and 72% in Nigeria and India respectively. More commonly than not, these women are amongst the poorest and most vulnerable; on the whole it is men who traditionally dominate the more lucrative and/or industrial fishery sectors[20] (Porter and Mbezi, 2010). Overall, women form a disproportionally large group among

[19]The Big Numbers Project was a joint activity of FAO, World Bank and WorldFish Center which aimed to provide disaggregated information on small and large-scale fisheries both at the global and country level (FAO&World Fish Center, 2008).

[20] Men more commonly fish with boats, particularly larger crew-manned industrial boats and are able to access the larger fish out at sea

the poor and vulnerable (Harrison, 2000). As much as 70% of the world's poor and 65% of the world's illiterate are female (IFAD, 2001; ILA, 1996; UNDP, 1995), and the vast majority of these women live in rural areas (UNDP, 2006).

FAO (2012) estimates that, overall, women account for at least 15% of the sector's primary workforce. Much of the fishing done by women within developing countries is for household subsistence needs and does not contribute to production figures. As such, the role women play within the fisheries remains largely invisible. For example, in many coastal countries women's near-shore and intertidal gleaning contributions go unnoticed and unreported in fishery statistics and policy documents (FAO and World Fish Center, 2008; Weeratunge et al., 2010). In fact, it is estimated that catch extracted by females account for approximately one-quarter of the total seafood harvested globally (Aguilar and Castaneda, 2001).

Women's contributions to artisanal fisheries, as well as the goods and income derived from these, affect not only the livelihoods of these women but also their households and wider communities. The gleaning of shellfish, urchins and clams from intertidal areas by women can represent the only sources of protein for vulnerable household members, including women and children; larger fish caught by the males are often sold on for cash income (Porter and Mbezi, 2010; Walmsey et al., 2006; Weeratunge et al., 2010). Evidence also suggests that in many fishing communities, it is the income from the women's activities which provide the bulk of a household's upkeep. In many fishing societies, male and female budgets are kept separate and a fisherman's income is his to spend (Bennett, 2005). In fact, women's income has been shown to have independent effects upon children's education, health and nutrition from that of men's. Moreover, the marginal effect of female income on child nutrition is a staggering four to eight times as high as that attributed to male income (Edmund, 2008). Nathan and Apu (1998) go as far as to report husbands reducing their contribution to household expenditures once they noted their wives and other female household members earning further income from new activities. Porter and Mbezi (2010) found that in two fishing communities along the Tanzanian coast the vast majority of fishing households were effectively female-headed households, dependent almost entirely on the wife's own subsistence activities, including fishing.

Lacking a voice within management decisions, women can further be displaced from established fishery roles if these fisheries become more commercialised and lucrative over time. For example, on Songo Songo Island, Tanzania, women have collected octopus from the intertidal areas for many years. However, as the price of octopus increased alongside an increasing scarcity in stock, men began to invade the fishery;

using boats to access deeper water, men now have the advantage, further depleting stock as well as preventing the stock from entering the shallower areas accessed by women (Porter and Mbezi, 2010). In general, fisheries are managed as open access or CPR. Successful management of CPR is based within local institutions and long-standing local rules and social norms (Ostrom, 1990). However, new patterns of income generation within these community property regimes (e.g. improved markets and/or efficiency of extraction) can frequently redefine the rules of resource access and control. This can result in new patterns of benefit distribution within both households and communities. In particular, the privatisation of previously communal areas through government and conservation policies can directly impact the livelihoods of women, placing previously accessible land into private ownership (Carney, 1993). Rural women rarely have legal or – in the case of many coastal areas – *defacto* control over natural resources; it is reported that women own less than 2% of titled land. As a result, women are often pushed further into the margins (OECD, 2001).

Social and cultural norms, educational opportunities and household commitments mean that women and men respond differently to development opportunities (Pandolfelli et al., 2008). Not only are women often ignored in fishery policy interventions, but what is worse, developments themselves can widen inequities and deprive women the established roles that once existed and were provided for them. Understanding these differences is paramount in moving towards fair representation of women within fisheries development policies and empowering women within coastal communities. Determining those factors which enable and promote female participation within novel policy tools, such as market-generating interventions which enviably any PES schemes will be, is paramount.

There are a number of prominent reasons why gender should be explored further within marine resource management and conservation initiatives, including marine PES schemes. These include: 1) women make up a large proportion of marine resource users; 2) possible omission and further marginalisation of poor community members with little voice; 3) omission and further marginalisation of chief child-carer and household provider and 4) exclusion of those members who rely most highly on natural resources. Instruments and policies which ignore the role of gender in resource management will serve to exclude, threaten the livelihoods and further marginalise an already vulnerable group as well as the wider community. Enabling and promoting the participation of women will serve to advance empowerment, development and successful resource management within coastal communities.

6.4 The case study

6.4.1 The Mtwara region

Despite efforts by the Tanzanian Government and major donors, 50% of the country's population live in poverty and average income is 16% below the national poverty line. Poverty levels continue to remain higher within rural areas (Ministry of Finance and Economic Affairs, 2010; Porter and Mbezi, 2010).

Like many other coastal African countries, recent years have witnessed increasing pressures upon coastal resources including illegal fishing practices, habitat destruction and growing populations (Sesabo and Tol, 2005). High and increasing poverty is prevalent amongst fishers: average yearly income in most Tanzanian coastal villages does not exceed US$ 100 per person; fish supplies per person are declining and excessive exploitation of the fishery continues (Cinner, 2010; Olale et al., 2010; Sesabo et al., 2006).

Located in the south of Tanzania, Mtwara region is considered among one of the country's poorest and least developed regions. Thirty eight percent of the population live below the basic needs poverty line, with the coastal population considered amongst the poorest (Guerreiro et al., 2010; Malleret, 2004). Coastal villages in the area show a high dependence on marine resources, as high as 63 – 74% of households in some; furthermore 54% of households directly depended on or were involved in fishing (Malleret 2004). These figures are consistent with other studies across Tanzania and Kenya (Malleret and Simbua, 2004).

In 1996, the number of registered fishers in the Mtwara region was estimated to be 2050, approximately 10% of Tanzania's total registered artisanal fleet; in 2010 this figure was more than double at 5,600 (Dadi 2010). This number is anticipated to be higher once non-registered male fishers and women are considered. With the rare exception, women fish using the gender-specific method 'tandilo' which involves dragging fine meshed nets (<1mm) along the shoreline at low or high tide. Tandilo was found to comprise 23% of all fishing methods within the surveyed groups (Malleret, 2004).

In 2010, CARE International, in collaboration with WWF, commissioned a study to investigate the possibility of a marine PES scheme to reduce community exploitation of fisheries for villages both within and outside of the marine park. As previously mentioned in Section 5.4.1 the Mnazi Bay area represents an important East African biodiversity hotspot and one of ecological importance for the surrounding marine areas, one of whose greatest threats comes from intensive and destructive fisheries

(WWF, 2004). The proposed PES design offers compensation in the form of cash to cover initial opportunity costs alongside training in alternative occupations for the longer term. Compensation is offered to mitigate the, at present hypothetical, closure of core marine zones within both the larger marine park and outside area. These core marine zones are identified relative to their biological significance, as determined in prior consultation reports[21].

More details on the case study, local demographics and the proposed marine PES programme can be found in Chapter 5 of this thesis.

6.4.2 Mtwara's fishing women

Women comprise just over half of Tanzania's total population (52%). On nearly all measures of health, education and economic status these women score lower than their men counterparts (Ministry of Finance and Economic Affairs, 2010; Porter and Mbezi, 2010). Overall, Tanzanian women are fairly inconspicuous within the formal fishing sector; men dominate the activity overall and occupy the more profitable sectors. However, within certain parts of the sector women play a significant role (Jiddawi and Öhman, 2002; Malleret, 2004).

The study focuses on six coastal villages located within Mtwara's two coastal districts: Mtwara Urban and Mtwara Rural. The study site is described in greater detail in Chapter 5 of this thesis. Within the coastal communities of Mtwara women are highly involved in tandilo fishing and gleaning activities. Recent household surveys by Malleret (2004) found tandilo fishing accounted for 23% of all fishing methods within coastal villages.

While men engage in many different types of fishing, women generally only partake in 'tandilo'. In the past 'tandilo' fishing involved catching small fish 'dagaa' from shore and timing depended on low and high tides. Tandilo fishing is normally conducted within intertidal areas close to shore, using mosquito nests sewn together. Three to six women drag a net parallel to the beach catching small fry fish (known locally as dagaa) and juvenile fish of other species. Women typically spend 2 to 4 hours fishing in this way during low or high tides, depending on the area. Daily catch rates are estimated at approximately 2-17 kg per woman (Jiddawi and Öhman, 2002; Malleret, 2004). Fishing is generally conducted for two weeks in each month. Women also involve themselves in gleaning molluscs from the shore as well as collecting sea cucumber and octopus during spring tides (Malleret, 2004).

[21] As described in Chapter 5. For more information see Samoilys (2010) and Yahya (2010)

Tandilo fishing is one of the more difficult fishing issues with which the marine park and fisheries officials have to deal with. This fishing method, which uses small meshing to catch small fish, is particularly destructive to the intertidal environment both via trampling of substrate and the catching of juveniles. Not perceived as damaging by the local communities, the majority of the catch (61%) is below half the maximum adult size (Jiddawi and Öhman, 2002). However, it can represent one of the only sources of income for these fishing women (Malleret, 2004).

6.5 Survey design and implementation

In order to analyse the determinants of willingness to participate in the potential marine PES programme, we use primary data from a household and stated preference survey conducted with 661 fishers located in Mtwara's two coastal districts. In particular we also look at the effect of risk mitigation strategies on shaping decisions to adopt the proposed PES scheme.

Questionnaire design followed the principles laid out by Bateman et al. (2002). Surveys collected data on: individual and household demographics; household assets; attitudes relating to fishing, the environment and conservation; fishing practices and income; diversification strategies of the individual and household, and social capital characteristics. A scenario was presented relating to the implementation of a possible PES programme. The survey provided information on the current situation as well the new scenario. Under the new scenario, specific core areas important as breeding and nursing sites in the locality of the surveyed villages would be closed (as described in Section 5.4 of Chapter 5 & Figure 5.3) and all illegal fishing would be terminated. The scenario further introduced the concept of a cash compensation scheme over a 4-year period with a further investment into alternative livelihoods. The level of cash compensation, although not offered as an initial bid at this stage, was presented as a value equivalent to current opportunity costs of participation. Information on implementing and regulating bodies was also provided. This was the combination of the on-ground NGOs (in this case WWF) and marine park authority within the Park and the on-ground NGO and BMU for those villages outside the park. Upon description of the new fishing scenario, fishers were given a choice as to whether they would be willing to participate in the marine PES scheme described. Those respondents who stated a willingness to participate were then asked for the required level of compensation, although these details are not included for the purpose of this analysis. The scenario was presented as on Section 5.6 of this thesis, page 134. The full survey is documented in Annex B1.

Male and female fishers are analysed separately, given both their differing production functions and also in interest of investigating those varying attributes in prompting participation across male and female fishers. It is important to note that while we are interested in reporting the differences between men and women, we are not attributing these inherently to "gender" per se but wish to say something about how those differences in opportunities, possibilities and livelihood strategies afforded to these women over their male counterparts can influence participation. Ultimately, whether it is a "gender" choice or a consideration of their livelihood options, the goal is to identify those attributes associated with participation.

We consider the presence of alternative occupations and social capital variables as a proxy for risk mitigation. Income diversification is measured as the number of alternative income generating activities at the individual level. These are further separated into agricultural and natural resource dependent and non natural resource dependent. Occupational diversity is also measured as number of alternative income and non-income occupations at the individual level as well as at the household level. We dissect social capital using four distinct indicators: known trust in others[22]; membership within a non-fishing group; and involvement within two social networks (bilateral dependency with others for fishing activities and bilateral reliance with others during times of fishing hardship[23]). Data pertaining to social inclusion was also collected within the surveys, however this was dropped from all analysis due to limited variation within the results: nearly all interviewees reported no involvement within decision-making at the village or resource use level.

The survey was implemented in the two coastal districts of the Mtwara Region described previously: Mtwara Urban and Mtwara Rural. Village selection was based on the prior work of Samoilys (2010) and Yahya (2010) which identified representative and appropriate villages based upon dependency on fishing as well as the depleting health of coastal resources[24]. Six coastal villages were selected: three within the marine park (Mngoji, Mkubiru and Msimbati) and three outside (Mikindani, Naumbu and Pemba). Focus groups and personal interviews with key informants identified relevant parties and shaped the design of the questionnaire.

[22] Not to be confused with 'Generalised trust' which measures an individuals expectation of others trustworthiness where decisions are based upon more general information about social groups and situations. Herein we address specific 'thick' trust whereby responses are based upon more first-hand knowledge of representative groups (Naef and Schupp, 2009; Newton, 2007)

[23] Bilateral dependency and reliance comprises three potential relationships: dependence upon other; depended on by other; or a combination of both.

[24] Further information as to selection can be found in Section 5.4.4 of this thesis.

Within these villages a sample of fishermen and fisherwomen were selected for personal structured interview. Initially, a random sampling technique was adopted: fishers were randomly selected from lists provided by local village leaders. However, the unpredictable nature of the fishers and their timetables resulted in fewer questionnaires than desired. This led to a more non-probabilistic method being utilised; additional fishers were identified and approached within villages and at landing sites. Piloting of surveys was conducted in the month of March and final questionnaires were collected from April through to June 2010. Of a total of 661 fishers interviewed across all six villages, 101 fished outside of the core zones and were therefore not eligible for inclusion within the PES programme and dropped from final analysis; a further 20 were also excluded because they were incomplete. The results below are based on the final usable sample of 540 fishers. In total 234 male and 306 female usable fisher surveys were collected.

6.6 Descriptive Results

Table 6.1 displays key demographic characteristics for the final sample of 540 fishers, and is further broken down to the village level and by gender. Patterns are fairly consistent across villages.

Average fisher age is 35 years and average village household sizes range from 4.5 – 5.6. In all villages, education levels were low: on average almost 40% claimed no schooling while the remaining majority held only some degree of primary education, only 2% claimed to have attended secondary education.

No significant differences were found between the mean demographic characteristics as broken down for male and female except fishing income. The largest disparity can be seen in male fisher earnings: Pemba averaged a fishing income of nearly US$ 7.5 per day for those days spent fishing giving an overall daily wage of approximately US$ 5.4. Fishing income within the male subgroup was found to be higher for those villages located outside of the marine park. This is possibly due to the higher incidence of deep-sea fishing in these villages, in particular Pemba village. As expected female fishing earnings were significantly lower than their male counterparts ($t=-7.746$, $p<0.001$).

Table 6.1 Mean demographic characteristics of sample respondents

	In			Out			All	In	Out
	Mkub	**Mngj**	**Msim**	**Mkdn**	**Naum**	**Pemb**			
No.	151	73	74	46	117	79	540	297	243
Male/Female	*56/95*	*27/46*	*28/46*	*33/13*	*53/64*	*37/42*	*234/306*	*111/186*	*123/120*
Female (%)	62.9	63.0	62.2	28.6	54.7	55.2	56.7	62.6	49.4
Age	35.2	35.5	36.2	44.2	33.9	30.6	35.3	35.6	34.7
Male/Female	*34.9/35.5*	*36.4/35.0*	*33.8/37.8*	*44.0/50.3*	*33.2/34.6*	*31.6/29.7*	*35.0/35.5*	*35.0/36.0*	*35.1/34.4*
HH_size	4.7	5.6	5.3	5.2	5.2	4.5	5.0	5.1	5.0
Male/Female	*4.5/4.8*	*5.4/5.7*	*4.8/5.7*	*5.4/4.8*	*5.5/4.9*	*4.4/4.5*	*5.0/5.1*	*4.8/5.3*	*5.1/4.8*
Education									
None	25.2	34.3	52.1	26.1	47.4	50.0	38.6	33.9	44.4
Male/Female	*26.8/24.2*	*37.0/32.6*	*46.4/54.5*	*30.3/15.4*	*50.0/45.3*	*47.2/52.4*	*39.2/37.9*	*34.2/33.2*	*43.8/45.0*
Primary	70.2	64.4	48.0	73.9	50.0	48.7	59.2	63.5	53.9
Male/Female	*62.5/74.7*	*59.3/67.4*	*53.6/45.5*	*69.7/84.6*	*48.1/51.6*	*52.8/45.2*	*57.3/60.9*	*59.5/66.3*	*55.4/52.3*
Secondary	3.3	1.4	0.0	0.0	1.7	1.3	1.7	2.0	1.2
Male/Female	*7.1/1.1*	*3.7/0.0*			*0.0/3.1*	*0.0/2.4*	*2.2/1.3*	*4.5/0.5*	*0.0/2.5*
Other	1.3	0.0	0.0	0.0	0.1	0.0	0.6	0.7	0.4
Male/Female	*3.6/0.0*				*1.9/0.0*		*1.3/0.0*	*1.8/0.0*	*0.8/0.0*
Fish income	1.67	0.99	0.97	2.23	1.55	3.08	1.72***	1.33	2.17
Male/Female	*2.65/1.02*	*1.61/0.66*	*1.11/0.90*	*2.72/0.89*	*1.96/1.22*	*5.37/1.11*	*2.66/1.00*	*2.06/0.90*	*3.17/1.14*

Where: In=villages located in park, Out=villages located outside of park; Mkub=Mkubiru, Mngi=Mngoji, Msim=Msimbati, Mkdn=Mikindani, Naum=Naumbu, Pemb=Pemba

Social capital and occupational diversification indicators also varied between villages and between men and women, a summary of which is presented in Table 6.2. From the table it can be also seen that women have a significantly higher number of alternative occupations than men, income generating or not ($t=2.481$, $p<0.05$ and $t=19.000$, $p<0.001$ respectively). However, male fisher households appeared to have significantly higher income generating alternatives than female fishers ($t=-2.605$, $p<0.01$).

Average trust levels were consistently high, averaging 3.9 out of a possible 5. Average trust was not seen to be significantly different for those fishers inside and outside of the park ($t=-0.431$). However, trust in authority was significantly higher for those located within the park at the 5% level ($t=-1.913$, $p<0.05$). Average trust in fishers from ones own village and other villages showed a reverse relationship; fishers within the park were significantly less trusting of fishers from within their village as well as from other villages ($t=2.045$, $p<0.05$ & $t=3.967$, $p<0.001$ respectively). Overall group membership outside of the fishing sector was fairly low, approximately 10% of interviewees. Reciprocal support networks averaged approximately 1.4 and 1.0 for

dependence to carry out fishing activities and reliance in times of bad fishing respectively. Fishers had, on average, one alternative income-generating source.

Table 6.2 Mean social capital and occupational characteristics of sample respondents

	In			Out			All	In	Out
	Mkub	Mngj	Msim	Mkdn	Naum	Pemb			
Social Capital									
Ave Trust	3.97	3.91	3.88	3.78	3.97	3.90	3.92***	3.93	3.91
Male/Female	*4.00/3.95*	*3.72/4.02*	*3.58/4.07*	*3.68/4.05*	*3.73/4.16*	*3.69/4.08*	*3.87/4.04*	*3.81/4.00*	*3.92/4.11*
Trust in Authority	4.27	4.30	4.23	3.88	4.15	4.12	4.19***	4.26	4.09
Male/Female	*3.94/4.46*	*3.85/4.57*	*3.50/4.67*	*3.47/4.92*	*3.42/4.75*	*3.46/4.68*	*3.62/4.62*	*3.81/4.53*	*3.45/4.75*
Trust fishers from village	4.42	4.32	4.12	4.45	4.49	4.47	4.39***	4.32	4.48
Male/Female	*4.71/4.24*	*4.41/4.27*	*4.21/4.07*	*4.65/4.00*	*4.66/4.34*	*4.71/4.27*	*4.60/4.23*	*4.51/4.20*	*4.67/4.29*
Trust fishers other village	2.69	2.91	2.43	3.17	3.26	3.33	2.93***	2.67	3.26
Male/Female	*3.33/2.35*	*3.45/2.52*	*2.40/2.44*	*3.86/1.61*	*4.13/2.52*	*4.29/2.26*	*3.65/2.37*	*3.12/2.40*	*4.11/2.32*
Group membership	0.17	0.15	0.12	0.15	0.02	0.00	0.10	0.15	0.04
Male/Female	*0.21/0.15*	*0.11/0.17*	*0.14/0.11*	*0.12/0.23*	*0.02/0.02*		*0.10/0.10*	*0.17/0.15*	*0.04/0.03*
Dependent on other for fishing	1.41	1.25	2.53	1.22	1.10	1.27	1.44***	1.65	1.19
Male/Female	*0.70/1.83*	*0.59/1.63*	*0.82/3.57*	*0.82/2.23*	*0.87/1.30*	*1.08/1.43*	*0.82/1.91*	*0.70/2.21*	*0.92/1.45*
Rely on other in bad times	1.19	1.21	1.00	0.72	0.79	0.68	0.96**	1.14	0.74
Male/Female	*0.82/1.40*	*0.78/1.46*	*0.61/1.24*	*0.48/1.31*	*1.02/0.59*	*0.78/0.60*	*0.78/1.10*	*0.76/1.38*	*0.80/0.68*
Presence of alternative activities									
Alt income	0.89	0.73	0.54	0.72	0.64	0.77	0.73**	0.76	0.70
Male/Female	*1.14/0.74*	*0.56/0.83*	*0.43/0.61*	*0.61/1.00*	*0.45/0.80*	*0.32/1.17*	*0.63/0.81*	*82.0/73.1*	*0.46/0.94*
Alt HH inc	1.28	1.08	0.66	1.07	1.05	1.00	1.06***	1.08	1.03
Male/Female	*1.96/0.88*	*1.07/1.09*	*0.86/0.54*	*1.03/1.15*	*1.08/1.03*	*0.73/1.24*	*1.20/0.95*	*1.47/0.85*	*0.96/1.11*
Nonfarm inc	0.89	0.62	0.54	0.85	0.91	0.87	0.80**	0.74	0.88
Male/Female	*1.48/0.56*	*0.56/0.65*	*0.68/0.46*	*0.82/0.92*	*0.85/0.95*	*0.62/1.10*	*0.91/0.73*	*1.05/0.55*	*0.77/0.99*
Alt activity	2.24	1.82	1.95	1.50	1.69	1.90	1.91***	2.06	1.72
Male/Female	*1.55/2.64*	*1.00/2.30*	*1.14/2.43*	*0.85/3.15*	*0.91/2.34*	*0.73/2.93*	*1.06/2.56*	*1.32/2.51*	*0.84/2.63*
Alt HH act	2.62	2.30	2.20	1.96	2.16	2.41	2.34***	2.45	2.20
Male/Female	*2.41/2.75*	*1.78/2.65*	*1.68/2.52*	*1.52/3.08*	*1.60/2.63*	*1.76/2.98*	*1.83/2.72*	*2.07/2.67*	*1.63/2.79*

Where: In=villages located in park, Out=villages located outside of park; Mkub=Mkubiru, Mngi=Mngoji, Msim=Msimbati, Mkdn=Mikindani, Naum=Naumbu, Pemb=Pemba *Represent results t-tests comparing means of male and female subpopulations. (*) denotes significance at the 10% level, (**) at the 5% level and (***) at the 1% level.

Comparison mean t-testing indicated significant differences between the genders for proxies of risk mitigation: social capital and occupational diversity attributes. Significance levels for mean comparison t-test results are indicated within Table 6.2. Of

the social capital characteristics only group membership was found to be similar for both men and women. Average trust levels were significantly higher in women for both average trust and trust in authority (t=6.033, p<0.001 & t=12.539, p<0.001 respectively). However, men displayed higher average ratings for trust in fishers from within their village as well as from other villages (t=-4.642, p<0.001 & t=-9.062, p<0.001 respectively). Women were also seen to have a significantly higher number of network linkages for dependency for fishing activities (dependent on others for fishing) and reliance in times of fishing hardship (rely on other in bad times), (t=8.018, p<0.001 and t=3.178, p<0.005 respectively). This is possibly due to the fact that tandilo fishing generally relies on three to six women dragging mosquito nets along the tidal zone. Within this framework it is also likely that these women support each other in times of trouble.

Willingness to participate in the proposed scheme is indicated in Table 6.3. Willingness to enrol varies both across villages and across gender within villages. Overall 60% of surveyed fishers were willing to enter into the hypothetical scheme. Willingness to participate varied across villages, with those outside of the park seemingly less willing to participate overall; only half of surveyed fishers located outside of the park were willing to consider enrolment vs. almost 70% within park boundaries. A t-test showed this willingness to participate to be significantly different at the 1% level (t=-4.824, p<0.001). Females were also significantly more likely to participate over men, this was significant to the 1% level (t=5.334, p<0.001). Again women within the park showed a slightly higher likelihood of enrolling, however, this difference was much smaller than was seen in their male counterparts, 71% vs. 68%; this difference was not significant (t=-0.6422, p=0.5212).

Table 6.3 Willingness to participate in proposed PES scheme

	In			Out					
	Mkub	**Mngj**	**Msim**	**Mkdni**	**Naum**	**Pemb**	**All**	**In**	**Out**
Willing to participate in PES scheme	74.8	54.8	71.6	47.8	47.0	43.0	60.0	69.0	49.0
Male/Female	*75.0/74.7*	*48.2/58.7*	*64.3/76.1*	*36.4/76.9*	*30.2/71.9*	*27.0/57.1*	*47.4/69.6*	*65.8/71.0*	*30.9/67.5*

6.7 Econometric strategy

The decision to participate in the PES scheme can be modelled as a dichotomous choice - a binary response – and the data collected indicates the observed choice but not the unobserved measure of 'relative attractiveness' of available options. Thus we utilise a

probit model where y is the binary dependent variable indicating fishers' decision to participate and y* is a latent variable measuring fisher's utility from their choice. X is a vector of explanatory variables affecting utility and ε is the error term with an assumed normal distribution with zero mean and variance σ^2.

$$y = \begin{cases} 1 & \text{if } y^* = U(x=1) - U(x=0) \geq 0 \\ 0 & \text{if } y^* = U(x=1) - U(x=0) < 0 \end{cases} \quad (1)$$

$$y^* = X\beta + \varepsilon \qquad \varepsilon \sim N(0, \sigma^2) \quad (2)$$

The probability of adoption is expressed in (3) where vector *x* is a vector of regressors including all demographic characteristics, *z* is a vector of regressors for all attitudinal characteristics and *k* is the vector of regressors representing income diversification and social capital characteristics. [*x* x *k*] denotes the vector containing the interaction term between demographic regressor location (Inpark) and social capital regressor average trust (Avetrust) and ε is the usual error term.

$$Pr[Y_i = 1] = \Phi(x_i'\alpha + z_i'\beta + k_i'\gamma + [x \times k]_i'\delta + \epsilon) \quad (3)$$

The first model analyses the associations between proxy risk mitigating variables and participation decision. In Model 2 we add variables for individual characteristics. In Model 3 we add assets (fishing and non fishing specific) and our final model, Model 4, individual perceptions and attitudes are included. Given the different relationship with local authorities and park enforcement for those living inside and outside of the park we insert an interaction term for village location and average trust (Models 4a & 4b). To take account of unobservable differences in variance between villages all models are clustered at the village level within the analysis. The variables (*X*), as determined in Chapter 4, are summarised in Table 6.4. We further attempt to control for any possible endogeneity that might arise through an additional village effects model which controls for unobserved heterogeneity at the village level[25] (results of this model, Model 5, are displayed in Annex B3).

[25] Although second to an instrument (instruments are notoriously difficult to identify within cross-sectional data), screening of the data identified no viable instruments to control for potential endogeneity of social capital variables.

Table 6.4 Variable list and descriptive statistics of independent variables

Variables	*Definition*	*Mean*	*s.d.*	*min*	*Max*
Male	Dummy for gender: male =1; female =0	0.4	0.5	0	1
Inpark	Dummy for location: village found inside park borders =1; village located outside =0	0.6	0.5	0	1
Age	Age of respondent (in years)	35.3	12.7	16.0	82.0
Education	Dummy for respondent's level of education: 1= attended secondary or above; 0 otherwise	0.0	0.1	0	1
Hhsize	Number of members in household	5.0	2.2	1	22.0
Fish_income	Continuous variable for respondent's daily income from fishing (US $), calculated as function of fishing income for days fished*annual fishing effort divided by actual days in year.	1.7	2.6	0	29.9
Own_boat	Dummy for those fisher's who owned own boat: 1=own boat; 0=don't own boat	0.2	0.4	0	1.0
Dhow	Dummy for fishing from dhow (a larger boat able to access outer reef): 1=fishes from dhow; 0=fishes from other or no boat	0.2	0.4	0	1.0
Legal	Dummy for those fishing using legal methods: 1=fish legally; 0=fish illegally	0.3	0.5	0	1.0
MSL	Material score index created from respondent household's assets. Index is calculated from presence of assets: 'high' quality of housing (roof and walls), ownership of transport vehicles and household appliances. Higher values indicate a higher asset wealth.	5.9	1.5	3	11.0
Land_area	Continuous variable for area of land owned	1.8	2.4	0	30.0
Attitudes					
Perceive_ben	Likert scale 1-5 for perceived change in number of fish caught in last 5 yrs: 1=a large decrease; 5=a large increase	2.2	1.3	1	5.0
Better_off	Likert scale 1-5 for perceived change in standard of living in last 5 yrs: 1=a large decrease	2.1	1.2	1	5.0
Cons_benefit	Likert scale 1-5 for attitude relating to potential benefit of marine conservation: 1=a large detriment; 5=a large benefit. Proxy for believes in conservation as beneficial	3.0	1.5	1	5.0
Happy_child	Likert scale 1-5 for attitude relating to feelings if son/daughter became fisher: 1=very unhappy; 5=very happy. Proxy for satisfaction with current fishing situation	3.0	1.5	1	5.0
Social capital and income diversification					
Alt_inc	Count variable for presence of alternative income	0.7	0.9	0	3.0
Alt_farm_inc	Dummy for presence of alternative cash activity from farming	0.2	0.4	0	1.0
Alt_nonfarm_inc	Count variable for presence of alternative non farm income	0.8	0.9	0	5.0
Alt_activity	Count variable for presence of alternative activity	1.9	1.2	0	4.0
Alt_hh_activity	Count variable for presence of alternative activity in household	2.3	1.1	0	10.0
Grp_mem	Dummy for respondent member of non-fishing group: 1=member; 0 = otherwise. Includes groups such as village community banking, church associations, livestock rearing groups.	0.1	0.3	0	1.0
Social_inclusion	Dummy for active participation in community decision making	0.0	0.1	0	1
Social_incl_marine	Dummy for active participation in decisions about community marine resource use	0.0	0.1	0	1
Dep_work	Count variable for number of dependency networks respondent is located within for fishing related activity corrected for village average, i.e. number of people respondent depends on and/or is depended upon by others to conduct fishing activity	1.0	1.1	0	4.1

Table 6.4 Variable list and descriptive statistics of independent variables (cont.)

Variables	*Definition*	*Mean*	*s.d.*	*min*	*Max*
Social capital and income diversification (cont)					
Dep_work_give	Count variable for number of networks whereby respondent gives assistance to others for fishing activities corrected for village average	1.0	1.3	0	4.9
Dep_work_rec	Count variable for number of networks whereby respondent receives assistance from others for fishing activities corrected for village average	1.0	1.2	0	4.1
Rely_hardtime	Count variable for number of networks respondent is located within for consumption smoothing corrected for village average, i.e. number of people respondent can turn to and/or is turned to during times of fishing hardship	1.0	1.2	0	7.6
Avetrust	Respondent's average level of trust. Average value of all trust variables. Continuous variable 1-5: 1 = no trust; 5 = fully trust	3.9	0.6	1.6	5.0
Avetrust_Auth	Respondent's average level of trust in authority. Average value of all trust variables for authoritative figures (includes trust in community leaders, local officials, fish enforcement officers) Continuous variable 1-5: 1 = no trust; 5 = fully trust	4.2	1.0	1.5	5.0
Trust_ fishers_vill	Respondent's average level of trust in other fishers in village. Continuous variable 1-5: 1 = no trust; 5 = fully trust	4.4	0.9	1.0	5.0
Trust_fishers_ nonvill	Respondent's average level of trust in other fishers from other villages. Continuous variable 1-5: 1 = no trust; 5 = fully trust	2.9	1.6	1.0	5.0
Interaction terms					
Trust*Inpark	Interaction term between avetrust and Inpark	2.2	2.0	0	5.0
TrustAuth*Inpark	Interaction term between avetrust_Auth and Inpark	2.4	2.2	0	5.0
TrustVill*Inpark	Interaction term between avetrust_fishers_vill and Inpark	2.4	2.3	0	5.0
TrustOther*Inpark	Interaction term between avetrust_fishers_nonvill and Inpark	1.5	1.7	0	5.0

We use Model 4 in final analysis of the results over the village fixed effects model (Model 5, Annex B3) for the following reasons. Firstly, the results across the models are consistent and the fixed effects model showed no great variation in parameter estimates from the simpler 'Inpark' model (Model 4). Secondly, the ease with which these two models can be interpreted varies dramatically, in particular the interpretation of the interaction term in a non-linear model[26]. Thirdly, PES schemes will need to function within marine parks as well as outside, hence in interests of analysis and policy relevance the authors are more interested in the differences between those

[26] Unlike in linear models the magnitude and sign of the interacted variables are not equal to the marginal effect of the interaction term in a non-linear model (Ai and Norton, 2003). In a non-linear model the interaction effect requires computing the cross derivative as the magnitude of this effect depends on all covariates within the model. Moreover the interaction effect can have different signs for different observations. For a more detailed discussion please see Ai and Norton (2003). For this reason we analyse the interaction effect using the approach described in Norton et al. (2004). This methodology is compatible with the presence of only one interaction term within the model.

villages located within and outside of the park verses the individual villages themselves. Lastly, in order to quantify the implications of the model on the decision to participate we compute the marginal effects, estimated at the sample means, for Model 4.

Male and female results are not pooled and modelled separately due to differing production functions and confirmed by the results of a Chow test (Annex B4). Female model specification is modified slightly allowing for differences in fishing type which show no variation within the female specification and are thus dropped from the analysis, e.g. all females fish from shore and so no 'dhow' heterogeneity is seen using tandilo an illegal method.

6.8 Regression results

The regression results are reported in Table 6.5 separated out for gender. A Chow test confirms the division of male and female models as appropriate (chi2 = 39.43, $p<0.0000$). The outputs from all models, specific to gender, are broadly consistent; results remain robust throughout. The analyses suggest that the various determinants differ in their influence on men and women. As can be seen in Table 6.5 and 6.6, the results show that different variables are influencing the choice of participation across the gender divide. A chow test, shown in Annex B4, indicate these determinants have significantly different effects across a number of variables. Characteristics, as previously identified in Chapter 4, are discussed below.

6.8.1 Individual characteristics

Age and household size showed no association with participation choice across either gender in the preferred models (Models 4, Table 6.5 and 6.6). Unlike in previous studies, older fishers were no less likely to adopt the proposed intervention.

Of the socio-demographic variables, interestingly, only education is associated with higher likelihood of participation for women at a significant level, albeit only at the 10% level. Women who reported higher levels of education were more likely to be willing to adopt the proposed scheme. Education was not seen to correlate with adoption decisions for male fishing counterparts and the implications of education was significant between sexes (Annex B4: Chow test).

Table 6.5 Estimation results: Male fishers willingness to participate

Variables	Model 1 SC	Model 2 + demo	Model 3 +assets	Model 4 +attitudes	Model 4 marginal effects	Model 4a In Park	Model 4b Out Park
Alt_inc	0.177* (0.099)	0.191* (0.070)	0.206*** (0.079)	0.239*** (0.037)	0.095 (0.015	0.234*** (0.052)	0.143* (0.080)
Grp_memb	0.178 (0.285)	-0.132 (0.220)	-0.088 (0.197)	0.034 (0.213)	0.013 (0.085)	-0.186 (0.345)	0.100 (0.145)
Dep_work	-0.007 (0.072)	0.062 (0.049)	0.129** (0.055)	0.030 (0.057)	0.012 (0.023)	-0.177 (0.386)	-0.022 (0.070)
Rely_hardtime	-0.102** (0.047)	-0.081* (0.048)	-0.067* (0.039)	-0.074 (0.047)	-0.029 (0.018)	-0.075 (0.206)	-0.049** (0.019)
Avetrust	0.261*** (0.046)	0.381*** (0.070)	0.402*** (0.044)	0.247*** (0.056)	0.098 (0.022)	0.119 (0.208)	0.373*** (0.086)
Trust*Inpark		-0.188*** (0.050)	-0.194* (0.116)	-0.126 (0.172)	-0.038† (0.021)	-	-
Inpark		1.626*** (0.275)	1.688*** (0.116)	1.056 (0.660)	0.402 (0.229)	-	-
Age		-0.011* (0.006)	-0.008 (0.007)	-0.008 (0.007)	0.003 (0.003)	-0.013* (0.001)	-0.011 (0.014)
Education		-0.171 (0.171)	-0.141 (0.199)	-0.168 (0.160)	-0.067 (0.064)	-0.545*** (0.056)	0.121 (0.191)
HHsize		0.056 (0.044)	0.084** (0.041)	0.069 (0.054)	0.027 (0.022)	-0.041* (0.023)	0.187*** (0.032)
MSL			0.033 (0.036)	0.071 (0.046)	0.028 (0.018)	-0.009 (0.113)	0.119** (0.056)
Land_area			-0.017 (0.015)	-0.029* (0.016)	-0.012 (0.006)	0.008 (0.033)	-0.045*** (0.014)
Fish_income			0.042 (0.043)	0.086* (0.047)	0.034 (0.018)	0.060 (0.119)	0.012*** (0.004)
Own_boat			-0.329 (0.235)	-0.416** (0.472)	-0.165 (0.070)	-0.495 (0.393)	-0.011** (0.237)
Dhow			-0.265** (0.107)	-0.321*** (0.121)	-0.127 (0.048)	-0.314 (0.212)	-0.497*** (0.064)
Legal			-0.140 (0.237)	-0.193 (0.203)	-0.077 (0.080)	-0.160 (0.552)	-0.398* (0.220)
Perceived_change				-0.285** (0.131)	-0.113 (0.052)	-0.717*** (0.177)	-0.035*** (0.009)
Better_off				-0.119 (0.117)	-0.047 (0.046)	0.204*** (0.059)	-0.325** (0.158)
Cons_benefit				0.187*** (0.052)	0.074 (0.021)	0.254** (0.119)	0.186*** (0.063)
Happy_child				-0.153** (0.077)	0.061 (0.031)	-0.303 (0.187)	-0.175*** (0.030)
_cons	-1.084*** (0.376)	-1.784*** (0.417)	-2.335*** (0.701)	-0.736 (1.001)			
N	232	231	223	223		103	120
LogLikelihood	-154.336	-138.761	-129.092	-116.439		-46.341	-58.200
PseudoR^2	0.0384	0.1320	0.1634	0.2454		0.2906	0.2119

Robust standard errors. */**/*** denotes significance at the 10/5/1% level respectively, Standard errors displayed in brackets. Village fixed effects model displayed and contrasted in Annex B3; †as reported by inteff function in STATA.

Table 6.6 Estimation results: Female fishers willingness to participate

Variables	Model 1f SC	Model 2f +demo	Model 3f +assets	Model 4f +attitudes	Model 4f marginal effects
Alt_inc	0.204***	0.221***	0.215***	0.245***	0.080
	(0.192)	(0.058)	(0.053)	(0.045)	(0.023)
Grp_memb	0.638***	0.758***	0.695**	0.672***	0.178
	(0.216)	(0.282)	(0.321)	(0.253)	(0.063)
Dep_work	-0.152***	-0.163***	-0.172***	-0.203***	-0.066
	(0.033)	(0.032)	(0.046)	(0.045)	(0.015)
Rely_hardtime	0.066	0.054	0.079	0.044	0.014
	(0.108)	(0.114)	(0.115)	(0.099)	(0.032)
Avetrust	-0.002	0.126	0.202	0.241	0.077
	(0.192)	(0.353)	(0.281)	(0.180)	(0.056)
Trust*Inpark		-0.213	-0.171	-0.299	0.086†
		(0.389)	(0.327)	(0.317)	(0.032)
Inpark		0.944	0.676	1.136	-0.380
		(1.500)	(1.401)	(1.182)	(0.375)
Age		-0.002	-0.001	-2.0e-04	-6.6e-05
		(0.004)	(0.004)	(0.003)	(0.001)
Education		0.161	0.268***	0.328*	0.107
		(0.122)	(0.104)	(0.177)	(0.062)
HHsize		-0.007	0.007	-0.016	-0.005
		(0.021)	(0.021)	(0.019)	(0.006)
MSL			-0.104	-0.087	-0.027
			(0.066)	(0.066)	(0.023)
Land_area			-0.020	-0.033	-0.011
			(0.052)	(0.044)	(0.015)
Fish_income			0.555*	-0.192	-0.063
			(0.303)	(0.320)	(0.106)
Perceived_change				-0.081	-0.027
				(0.156)	(0.023)
Better_off				-0.234***	-0.077
				(0.054)	(0.019)
Cons_benefit				0.184**	0.060
				(0.074)	(0.023)
Happy_child				-0.209***	-0.068
				(0.054)	(0.019)
_cons		-0.086	0.375	-0.890	-0.890
		(1.421)	(1.577)	(1.271)	(1.271)
N		296	286	286	
LogLikelihood		-170.859	-162.984	-147.787	
PseudoR^2		0.0560	0.0724	0.1589	

Robust standard errors. */**/*** denotes significance at the 10/5/1% level respectively, Standard errors displayed in brackets. Village fixed effects displayed in Annex B3; †as reported by inteff function in STATA.

Determinants of male willingness to participate were more closely associated with fishing income and investment into the sector, for example: owning a boat. Working on a dhow or owning one's own boat was strongly associated with non-adoption. In fact, these variables displayed two of the highest partial effects. Within the model, boat

ownership was associated with a 16.5% higher probability of rejection, working on a dhow showed a similar effect with a slightly lower probability of 12.7%. However, interestingly enough, fisher type (here analysed as fisher's using legal or illegal gear) showed no significant association.

Interestingly, one's MSL score displayed no significant relationship with participation choice in the preferred model; the p-value fell a little outside of the 10% level for male participants. This said, within the male village fixed effects model (Model 5A, Annex B3) MSL score displayed a positive association at the 10% level, whereby male fishers showing higher asset scores indicated a stronger likelihood of adoption. A t-test shows no such relationship was seen for female fishers (Annex B4: Chow test).

Surprisingly land size showed a negative relationship with adoption decisions in male fishers, i.e. male fishers with larger landholdings were less likely to adopt the marine PES. However, no such association was seen for female fishers and a chow test (Annex B4: Chow test) was unable to determine a significant difference between the two.

6.8.2 Individual environmental beliefs and attitudes

Attitudes displayed a strong association with participation likelihood, and appeared to show some variation in influence across the sexes, although we are unable to statistically prove this (Annex B4: Chow test). Viewing conservation as beneficial was a significant positive determinant for both sexes and the only consistent attitudinal predictor across the two models at the 1% level. Fishers of both sexes were also associated with a lower likelihood of signing up if they expressed happiness for their child to follow in their fishing footsteps, this association was more strongly significant for women at the 1% level than men (at the 10% level). Women were also less likely to be willing to participate in the proposed scheme if they had perceived an improvement in the standard of their living over the last five years. In a similar vein, male fishers who reported a positive change in the number of fish caught over the last 5 years were associated with a lower likelihood of participation.

6.8.3 Individual perceptions of risk and vulnerability

6.8.3.1 Income diversification

With respect to social capital and occupational diversity, only one variable was shown to be a consistent predictor across gender: those with alternative income activities were consistently associated with a higher likelihood of participation in the proposed PES scheme (Table 6.5 and 6.6, Model 4). This result was seen for both sexes.

The marginal effect of having an additional alternative income was 0.095 and 0.080 for male and females respectively. Or in other words, holding all other variables at the mean, the presence of an additional income activity was associated with a 9.5 percentage point increase in probability of adoption for men and an 8.0 percentage point increase in women. Having one alternative income source (Alt_inc=1) vs. none (Alt_inc=0), holding all other variables at the mean, resulted in a predicted probability of 50.3% vs. 40.8% for males and 75.2% vs. 66.9% for females.

As discussed previously, the literature suggests that some alternative occupations are more favourable over others to spread risk, and as such, different income activities could have differential effects upon participation choice. In order to investigate this, we reran the analyses for alternative model specifications substituting individual alternative income activities (Alt_inc) with individual farm income (Alt_farm_inc) and non-farm income (Alt_nonfarm_inc) separately[27]. Replacing Alt_inc in Model 4 (Tables 6.5 and 6.6) with these variables gave no significantly different outputs, and both income types were strongly and positively correlated with participation. Again substituting Alt_inc in Model 4 (Tables 6.5 and 6.6) with Alt_activity also gives similar results within the model[28].

6.8.3.2 Group participation

Membership within a non-fishing group (Grp_memb) displayed a very different association with participation choice for men and women.

For female fishers, this attribute emerges as one of the most influential, with a marginal effect of 0.178 (Table 6.6, Model 4); or in other words, the presence of a female fisher within a non-fishing group increases her probability of participating in the PES scheme by 17.8%.

Group membership, however, was not seen to be significant for men; a chow test indicated coefficients for males and females to be significantly different at 10% level (Annex B4: Chow test). This differential result suggests that membership within groups and their associated activities have a greater influence on female willingness to participate.

[27] Model results are not presented within this paper but are available from authors upon request

[28] Again model results are not presented within this paper but are available from authors upon request

6.8.3.3 *Dependency networks*

Much like membership in a non-fishing group, presence within a fishing dependency network indicated a significant relationship with adoption for women but not men. However, and of particular note, while the presence of alternative income sources and group membership indicate positive associations for these females, presence within a fishing dependency network showed a very different relationship.

Women who were associated with fishing dependency networks (Dep_work) appeared to limit their engagement in the proposed PES. There was, however, no significant variation in mean dependency networks across the wealth quartiles (ANOVA T=-0.607, P=0.544). Significant at the 1% level, involvement within fishing dependency networks was associated with a reduced likelihood of signing up by a marginal effect of -0.066. In contrast, for men this relationship was not seen to be significant and a chow test showed coefficients to be significantly different at the 1% level (Annex B4: Chow test).

To analyse this result further, separate analyses were conducted[29] in order to look at each side of the fishing dependency network: that is, we separately analysed all those who are depended upon by others to conduct fishing activities and all those who depend on others[30]. The relationship towards participation was not seen to change. However, it proved extremely difficult to disentangle which aspect of this network was holding people back from participation. This is perhaps because few interviewees were seen to have only one-directional networks, whereby they were seen to only depend on others to conduct fishing or only were only depended on. In fact, only 2% of female fishers reported receiving assistance only and 1% giving assistance only.

6.8.3.4 *Trust and location*

Given the different experiences of those villages within the park and those outside relating to enforcement and local authorities, an interaction term between trust and location was examined (Trust*Inpark). Looking at the male model (Models 4a & 4b, Table 6.5), the inclusion of the interaction term is further warranted after analysis of subpopulation models for in and out of park (see Models 4a & 4b, Table 6.5) which display a significant association for the trust variable for those outside of the park[31]. Average trust enters as significant and positive for those male fishers located outside of

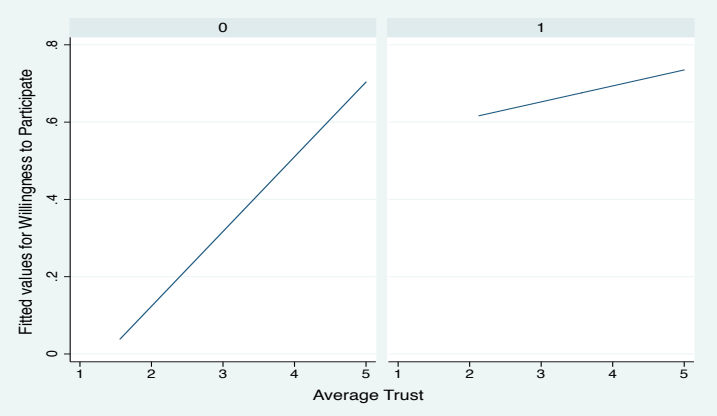
[29] Analysis not shown here but available from the authors on request

[30] Two additional models were run: Dep_work was substituted with Dep_work_give or Dep_work_rec

[31] While the addition of an interaction term does little to improve the model fit, the extra term absorbs much of the variation in the data, as we would expect from the subpopulation models estimated earlier.

the park boundaries, but shows no significant association for those located within. In the pooled model, the interaction term enters negatively but is not significant. As a result, we are unable to reject the hypothesis that the impact of trust is significantly different for those inside the park. However, that the association is significant for those outside of the park merits the inclusion of the interaction term. Plotting the predicted probabilities shows the marginal effect of trust on those within and outside of the park. Graphs are displayed in Figure 6.1.

Figure 6.1 Fitted probabilities for willingness to participate against average trust (avetrust) for subgroups inside (subheading 1) and outside of park (subheading 0)



Overall, average trust was the only social capital variable found to be associated with male participation, and yet was one of only two to not be significantly associated with adoption in female fishers. However, we are unable to say with confidence that the coefficients were significantly different (Annex B4: Chow test). One might wonder if the difference seen between the sexes with respect to trust might be associated with their very different production functions and the regulation they face. Given the very different production functions faced by both male fishers and female fishers it was not possible to control directly for this. However, if trust were to be associated with regulation (vs. gender) one might anticipate those male fishers who utilise illegal fishing gear to be less trusting. However, this was not seen to be the case. Moreover, a t-test on illegal fishers showed females to have significantly higher levels of trust than their illegal male fisher counterparts ($t= -2.5537$, $p<0.006$).

Within the male models, average trust (Avetrust) is seen to be a significant positive determinant for those located outside of the park. Although at first glance the coefficient for the location variable Inpark appears insignificant in Table 6.5 Model 4, the impact of location must be interpreted as the sum of the main effect and the

interaction term. This impact is therefore found to be statistically significant (p=0.06)[32]. Hence location also displays an association with adoption decision.

As such, location now (Inpark) appears to exert a large significant influence on participation within the model, however again the inclusion of the interaction term (Trust*Inpark) means interpretation is not as simple as looking at this main marginal effect (0.402, Table 6.5, Model 4 marginal effects). In practice, the marginal effect is calculated as 0.228[33] at the sample mean; those living within the park were 22.8% more likely to participate all other things held constant.

Individual trust categories are further analysed for male fishers in order to better interpret possible patterns. Regressions were rerun substituting Trust in authority (Avetrust_Auth); Trust in other fishers from village (Trust_fishers_vill) & Trust in fishers from other villages (Trust_fishers_nonvill) as defined in Table 6.4. Summary results are displayed in Table 6.7 below. Full model specifications are in Annex B5.

Table 6.7 Estimation results for varying measures of trust: Male fishers

Variables	Model 4 Average Trust	Model 6 Trust in Authority	Model 7 Trust in fishers from village	Model 8 Trust in fishers from other villages	Model 6b Trust in Authority marginal effects
Alt_inc	0.239*** (0.037)	0.243*** (0.047)	0.288*** (0.034)	0.359*** (0.094)	0.097 (0.019)
Grp_memb	0.034 (0.213)	0.048 (0.213)	0.092 (0.231)	0.123 (0.231)	0.019 (0.085)
Dep_work	0.030 (0.057)	0.048 (0.060)	-0.007 (0.068)	-0.021 (0.084)	0.019 (0.024)
Rely_hardtime	-0.074 (0.047)	-0.080* (0.045)	-0.082* (0.047)	-0.083* (0.048)	-0.032 (0.018)
Trust[#] Variable	0.247*** (0.056)	0.103* (0.058)	0.376*** (0.143)	0.110 (0.166)	0.041 (0.023)
Interaction term for trust*InPark	-0.126 (0.172)	-0.104 (0.116)	-0.251 (0.219)	-0.155 (0.184)	-0.041 (0.046)
InPark	1.056 (0.660)	0.955** (0.439)	1.706 (1.039)	1.131** (0.574)	0.337 (0.156)
_cons	-0.736 (1.001)	-0.182 (0.940)	-1.436 (0.938)	-0.085 (1.271)	
N	223	223	221	199	
LogLikelihood	-116.439	-116.754	-113.812	-99.938	
PseudoR^2	0.2454	0.2433	0.2561	0.2748	

Robust standard errors. */**/*** denotes significance at the 10/5/1% level respectively, Standard errors displayed in brackets [#]Trust variable represents 'Trust in Authority'; 'Trust in fishers from village' & 'Trust in fishers from other villages' in models 5, 6 & 7 respectively as well as for subsequent interaction terms.

[32] Calculated from chi2 of Inpark+Trust*Inpark=0 in STATA

[33] Calculated from margins function in STATA.

As is seen in Table 6.7 the respective trust variable has a positive effect on willingness to participate, irrelevant of location, for Models 4, 6 & 7. Trust in authority and trust in other fishers from village show a positive association with participation and can be interpreted in a similar manner as average trust (Avetrust).

Interestingly, however, trust in fishers from other villages shows no significant association with participation choice, in fact p is seen to be >0.5; whilst location retains significance.

6.9 Discussion

Many of the findings herein align with adoption literature discussed in the previous chapter and the subsequent stated hypotheses. However, what is apparent from the results presented is that these can be very different for male and female participants. In addition, results suggest that variables commonly associated with risk mitigation strategies, such as income diversity and social capital, may have implications for PES participation. Again, these may vary for both male and female fishers. At present there is little segregation within the adoption literature as to gender variation.

6.9.1 Determinants of fisher adoption

6.9.1.1 Individual characteristics and attitudes

The results of this paper suggest that men and women appear subject to different motivations when considering participation in the proposed marine PES scheme. Education, which is commonly shown as a positive determinant of adoption and participation choice in conservation agriculture as well as PES schemes, was only seen to influence females' participation herein. Male participation was seen to be more closely related to socioeconomic variables such as fishing income and investment into the sector. For example, working on a dhow or owning one's own boat were strongly associated with non-adoption, and displayed two of the highest marginal effects of all male attributes.

Of the individual characteristics, only age and household size revealed no associations across both sexes. Although we hypothesised decreasing participation with respect to increasing age and household size, the adoption literature remains unclear as to their role as determinants with many studies finding significant positive and negative influences and some reporting none at all (Chen et al., 2009; Knowler and Bradshaw, 2007; Mercer, 2004). In keeping with the literature, individual attitudes displayed strong associations with PES adoption, again with some variation between male and female fishers.

6.9.1.2 Individual perceptions of risk and vulnerability

Only one variable, occupational diversity was seen to positively influence both male and female fishers. The presence of an alternative income activity (Alt_inc) within a fisher's portfolio was a consistent determinant of participation, and is in line with current thinking (Chen et al., 2009; Deressa et al., 2009; Zbinden and Lee, 2005). Possibly, experience with alternative occupations, particularly income generating occupations, provide fishers with alternative skills and experience which allow them to more easily and more comfortably branch away from fishing. In addition to the possibility of gained experience, those with alternative activities would likely gain more from the PES scheme as in addition to payments, these individuals can increase monetary and time investments into these alternative activities; this may be particularly true of income generating activities. In a similar vein, those who are aware that they rely solely on fishing activities may be more risk adverse towards new fishing conservation schemes; more often than not these interventions promoted increased restrictions as well as a reduction in fishing effort. However, one might wonder if the presence of alternative livelihoods is merely encompassing the preferences of those less frequent, part-time fishers, hence those more willing to exit the fishery or reduce effort. If this were the case, one would expect fishing income (calculated as a function of daily income and annual fishing effort) to be negatively correlated with willingness to participate. However, fishing income shows a positive correlation with willingness to participate, albeit it at the 10% level for male fishers only, implying that the presence of alternative occupations has an effect on participation independent of 'fisher effort'.

Results for alternative income sources not differentiated for type are consistent with results when variables are differentiated for farm, non-farm and business incomes. This implies that, although some activities may be more fruitful than others, in respect to PES participation it is the occurrence and perhaps experience of other successful alternatives, rather than the alternative itself which aids participation.

This is where the similarities end. With respect to social capital, where men appear swayed by improved trust, a woman's choice is associated more by her current networks and group participation. And in fact for women these two characteristics may be in conflict.

For male fishers, trust was seen to be the only significant social capital variable predicting participation; trust can buffer the risk fishers perceive in participating (Mariola, 2012). For fishers to engage with a PES, they often have to admit that fishing efforts are too high and or are destructive in nature, and that they are a contributing factor. As a result fishers can open themselves up to increased scrutiny; imposition of

new regulations; and increased institutional oversight and a possible loss of autonomy related to operations (Breetz et al., 2005; Langpap, 2004; Mariola, 2012). Quite simply, the higher a fisher's trust the more he believes the PES scheme will function over the long-term through trust in the information provided about the scheme, faith that undesirable effects will be mitigated and belief in the payment itself (Kabii and Horwitz, 2006). In rural Mexico, Kerr et al. (2012) also found low levels of trust in village authorities to reduce participation rates in pro-social community tasks only in those instances where funds were channelled through said authorities. Interestingly, though, the influence of trust varied across location and gender.

The association of trust with participation was seen to vary between villages located inside and outside of the park, displaying a much smaller marginal effect for those inside of the park.

One possible reason for this difference within the park may result from the more intense regulation and enforcement which they face; this additional enforcement serves to reduce the relative value from fishing (Robinson et al., 2012). Although legally, fishing laws and regulations are identical within the marine park and outside, fishers within the park have more contact with patrol officers and experience more intense regulation. Therefore, although trust may be a strong predictor of participation under more typical circumstances, for those fishers residing within the marine park boundaries other institutional and regulatory issues override this. For these fishers within the park, reversing park implementation is not an option; Having experienced 'more severe' restrictions since the park's gazetting in 2000 these fishers may feel like cooperation is the only option, more so if they believe more severe enforcement and restrictions in the future. On the other hand, for those fishers outside of the park who experience lower rates of enforcement may wish to prevent additional forms of regulation, whether they be enforced by governments or other institutions; more so when trust in these institutions and others is low.

Moreover, female fishers with higher levels of trust displayed no more likelihood of PES adoption. Initially this may seem counter intuitive, yet trust also expresses belief about future actions and can represent a 'leap of faith' (Padmanabhan, 2008). Fishers may be uncertain as to how restrictions will change and ultimately increase in the future. To some degree, female uncertainty is lower: tandilo fishing will be fully restricted under the predefined PES.

Indeed, similar results were seen for trust measures in authority and village fishers. That trust in authority displayed a significant and positive relationship again seems intuitive and is aligned with the literature: lack of trust in regulating bodies has been

shown to prevent adoption of environmental initiatives all together (Moore et al., 2008; Parker et al., 2009). More specifically, a number of studies found mistrust WQT regulators to reduce initial adoption in the PES-like schemes (Breetz et al., 2005; Mariola, 2012). With respect to protected areas, local residents with higher degrees of trust for protected area officials displayed less opposition to protected area implementation and exercised higher levels of compliance (Stern, 2008).

However, trust in other fishers from village also indicated a positive association. One might wonder if this is simply an underlying trust, more generally, which is being caught here, however trust in other fishers from villages and average trust overall show little correlation. Moreover, trust in fishers from other villagers did not display a significant association with participation. Given the required participation and compliance of such fishers within a possible marine PES scheme, this is an interesting result. Fishers may also be considering the actions of others in their decision to participate, a reasonable consideration given that the 'cheating' of others will affect PES outcomes and success. However, this trust seems limited to those closer, more immediate fishers. Perhaps simply, these fishers do not consider these more distant players when thinking about the PES implementation. Indeed, Polman and Slangen (2008) also find that more generalised trust had no significant effect on participation choice. Or alternatively, they may believe enforcement will play a key role. However, this later reasoning seems hard to believe given the current enforcement success rates and high levels of illegal fishing.

If one were to look at location alone, the marginal effect would seem large (e.g. whether a village is located within the park or outside). In fact this variable shows the largest marginal influence at 0.228. While an interaction term was entered in order to tease out any influence on varying interactions with enforcement officials and authority relating to the park, it is likely that the presence of the park itself is also having additional influence on willingness to participate. Indeed, those outside may be more resistant to new restrictions of any kind, whether it be a PES scheme or not, whereas those within have become more desensitised to change.

For women, the picture was a little less straightforward. While a male fisher within these communities was swayed by improving trust, a female's choice was more closely associated with her current networks and group participation. And, these two characteristics were, in fact, in conflict. Whereas the results from alternative income generating activities and group membership confirmed current thinking in promoting adoption decisions (Chen et al., 2009; Deressa et al., 2009; Knowler and Bradshaw,

2007), presence within a dependency network actually correlated with a reduced likelihood of participation.

Group membership, along with education and alternative income generating activities were shown to have positive and significant association with participation in female fishers. The presence of these as positive predictors of willingness of participation may indicate that those women with a greater experience and/or confidence in working with others, and overall, may more readily engage in novel activities. Indeed, previous work has also shown that membership makes women more self-confident, assertive and vocal in gatherings, including within mixed contexts (Agarwal, 1997; Giri and Darnhofer, 2010; Pokharel et al., 2009), and that women may benefit disproportionally to their male counterparts from membership (Godquin and Quisumbing, 2008). However, it should be noted that while previous works confirm such thinking, interpretation must be done with caution; it is also possible that more confident and assertive females involve themselves in groups in the first instance. The data herein, and a lack of any instrument, does not enable separation of the two effects.

However, rather unexpectedly, one social capital variable was shown to have a negative association with female adoption; incidence within a fishing dependent network could hold her back. This reciprocal dependency relationship appears to lock women fishers in to their current status quo and dissuade participation in the PES scheme. This could be due to a number of factors.

One line of thinking is that fishers often become indebted to local businessmen who loan equipment and/or bail fishers out in times of hardship and these are difficult to exit. However, being located within a network which provided aid in times of bad fishing (Rely_hardtime) showed no significant relationship with participation, nor was this relationship seen to be significant for male fishers, who are more often those fishers indebted to businessmen.

Women are often involved in many complex networks and alliances to enable them better access to these resources (Bennett et al., 2004). Evidence from across a wide range of countries suggests that females of low-income groups are often those with the strongest social and kin ties (Molyneux, 2002). These women whom rely on strong networks and acts of reciprocity may be less eager to shift the balance and experiment with new initiative, due to reluctance to disturb these existing safety nets; the benefits of the reciprocal relationships could extend further than fishing alone.

Indeed, these networks may form an underlying base for insurance and self-protection mechanisms, as have been demonstrated in some Ethiopian farming systems (Di Falco

et al. 2010). Alternatively, female fishers may place a value on these networks beyond what is captured herein. Again, that a similar relationship was not seen for those relationships which provided assistance during bouts of bad fishing (Rely_hardship) implies some intrinsic value beyond an insurance mechanism against economic loss, but associated more with the fishing network.

As such, these traditional reciprocal sharing or assistance norms, while beneficial in many ways, under some circumstances can be injurious to household development (Di Falco and Bulte, 2011). It is worth noting here that under the local conditions found within the study, we assume a PES scheme to be a financial improvement to individuals, freeing up time and opportunity to explore alternatives. However this may not always be the case or at least not perceived as so.

This said, we anticipate that this result is applicable to not only to PES schemes, but also other conservation schemes and development activities. More qualitative and quantitative work is needed to verify this.

Understanding how women value these alliances will be important in determining how to induce female participation, or indeed how to design programmes which do not erode these networks.

6.9.2 Limitations

Given the cross-sectional nature of our data we are unable to tease out any causal relationships and can only point to associations within the data.

Although variables were selected with careful consideration of the literature, social capital by its very nature is difficult to measure and possible issues of endogeneity and omitted variables must be acknowledge. Further difficulties in identifying any relevant instruments also prevented demonstration of causal links and means the existence of endogeneity bias cannot be completely ruled out. Where possible, methods such as clustering for village and robustness checks against village fixed-effects models were employed to reduce endogeneity bias.

The cross-sectional nature of the data also fails to address any possible dynamic relationships within these PES schemes, for example changes in income, those potential of actors who initially take up the scheme but drop out at a later date or indeed those who initially hold back but enrol at a later date; this later group may be of significant interest given a perhaps less risky association with an established programme. That surveys were conducted during the months of April through June also means that results may be a seasonal snapshot only.

Further limitations worth mentioning are those associated with all stated preference studies. The hypothetical nature of the participation calls into question whether respondents are answering the question honestly given that there are no real life repercussions. As with all other such stated preference methods, the validity of the results rely on having an accurate, meaningful and understandable scenario, and perhaps to a greater degree a believable vehicle with adequate sanctions (Bateman et al., 2002). Careful design and piloting was undertaken to reduce all hypothetical bias and lead the authors to believe the results to be a good representation of reality.

Efforts were also made in order to gather data from a representative sample. In the first instance a random sampling method was adopted, however given the difficulty in locating pre-determined fishers a non-probabilistic method became necessary. Non-probabilistic sampling methods can lead to self-selection and sampling bias. However, previous studies in the area and in Tanzania more generally indicate a good representative sample overall and a lower likelihood of sampling bias.

Despite the limitations associated with our data, this study highlights possible linkages and barriers that various forms of social capital may play in PES uptake. However, one must note that this work is a starting platform from which other research is needed; it will be important to repeat the current analysis over a time series, allowing use of instrumental variable to reduce the possibility of endogeneity, examining those dynamic issues and move beyond a hypothetical context.

6.9.3 Policy Implications

The results presented here have interesting implications for the development of marine PES schemes, possibly of PES schemes more generally, and in particular those underlying conditions required to facilitate their development in the first place.

Overall, social capital variables showed a positive association with participation, although these varied between male and female fishers. Building trust and group participation can be seen as important prerequisites to any PES scheme. Furthermore, diversification of livelihoods – a common feature of many natural resource conservation schemes – should not be overlooked in PES design. In fact, PES should actively support their presence, particularly when a PES scheme calls for a reduction in fishing effort or indeed any natural resource harvesting. It is also important that PES interventions do not overlook past lessons from previous income diversification interventions. As in other marine conservation interventions the presence of alternative occupations may promote long-term success (Pomeroy et al. 2001). What is more, PES can call for conditionality when running alongside livelihood diversification which

previous conservation initiatives have been unable to do. For example, a PES scheme will automatically set a level of allowable exploitation or set aside which previous conservation and development schemes investing into alternatives were unable to regulate.

However, not all forms of social capital emerged as conducive with the PES scheme. As such PES programmes need to fully understand how networks and more specifically 'reciprocal dependency relationships' can influence willingness to participate. Particularly as such reciprocal relationships have been shown to play greater importance within poorer households and communities as well as for women (Dercon, 2002; Molyneux, 2002; Wakefield and Poland, 2005). The implications of this can have serious consequences if PES schemes do indeed wish to improve the livelihoods of those poorer more marginalised groups.

That different attributes, including social capital variables, were seemingly more or less important for male and female fishers as discussed above is significant, yet within the adoption literature there has been a focus on pooled data. Why is this important, one might ask? For two reasons, the first is clear-cut, PES should not under-represent or further marginalise women; determining which attributes may motivate participation, and indeed those barriers which can prevent it such as those networks based on reciprocity, is a necessary step in avoiding this.

Moreover, males and females are known to hold differing views towards natural resources and income. Women's participation in natural resource management continues to be regarded as pivotal to the sustainable use and management of resources (Mwangi et al., 2011; Resurreccion, 2006). Evidence even suggests that women hold values which make them more likely to promote environmental protection (Agarwal, 2000; Agrawal et al., 2006; Westermann et al., 2005) as well as invest in household development (Edmund, 2008). Despite this, policy design continues to overlook barriers to female participation, particularly within the fishery sector where women remain invisible due to their low income-generating power (Bennett, 2005; Harper et al., 2013). In some instances, targeting women may represent a win-win scenario.

Indeed in our case study, women were significantly more likely to sign up for the marine PES scheme than their male counterparts, even though implementation of said PES would result in a total ban on tandilo fishing. This form of fishing also represents one of the more destructive types, removing many juveniles and destroying substrate through repeated trampling.

Moreover, these female fishers are perhaps more ready and happy to replace fishing with another income generating activity; for these women it may be less about the tradition of fishing per se. Here, working with women fishers could represent a win-win: high environmental returns with low resistance.

However, women's fishing activities also tend to fit within a wider household portfolio, unlike men who are often the 'full time' fishers. And PES instruments will need to acknowledge and work around the part-time nature of female fishing activities and further household time constraints faced by women will more likely induce participation. Identification of appropriate alternatives to fishing need to be investigated, and these, as shown by Peterson & Stead (2011), can further have a gender component.

Targeting women will rely on studies which identify their motivations and constraints, and in designing policies which do not erode pre-existing relationships and which are in line with their needs.

Studies need to further identify additional determinants of female participation as well as the possible effects social networks may plan in female adoption decisions. The fact that an association is seen within this paper indicates that further investigation is warranted, both in how these can influence adoption and also in what implications the implementation of PES may have on these pre-existing networks. Furthermore, it would interesting to examine other elements of social capital not discussed here and in particular if presence within other networks display similar or even opposite patterns. Another key issue for women within PES schemes which should not be overlooked is the ability of women to keep control of resources; this is particularly important in a marine setting where *defacto* and common property rules are common.

More generally, it is a mistake to assume all women within communities are equal. To say all women share the same desires and/or needs is a gross oversimplification. Female participants may more closely align their needs with those of their kin than other women in general. Indeed, those women involved in the design and management of said schemes may reinforce the exclusion of others (Cornwall, 2003). Identifying what, if any, common determinants exist can perhaps help identify those females who continue to be disregarded. Non-fishing women also exist within coastal communities – perhaps to a greater degree than men – and these can represent poorer women within society, i.e. those relying on subsistence farming alone and without any form of individual income. These women will automatically be omitted from participating, the implications of which are unknown. Women also hold other related roles within the sector, as traders or processors. Although not directly addressed with

a marine PES scheme, these women will also be affected by changes in production or reconfigurations within the sector.

However, studies should not be limited to the study of women alone. Fishing communities have rich social structures and female empowerment cannot be achieved solely through female-targeted development interventions. Understanding the social structure of male-female relationships, rules, roles and rights is paramount to PES design. Indeed as women increase their incomes, how will husbands react? Past studies have shown that fisher husbands may even reduce contributions (Nathan and Apu, 1998). Furthermore, by 'privatising' CPR, PES schemes need to be aware of the consequences, particularly on the female and poorer members of society. In the past, interventions which privatise and bring income-benefits have served to further marginalise those lacking a voice or the power to control resources. As PES fundamentally serve to 'pay' resource-users for improved environmental management, schemes very much need to scrutinise what consequences may arise and how to mitigate these. PES must learn from these past mistakes in order not to repeat them in the future.

6.10 Conclusions

PES schemes continue to attract interest from policy makers, conservation practitioners, development practitioner and communities alike. What makes them so attractive to so many is their potential to pursue objectives beyond conservation, in particular that of regional development and poverty alleviation. However, this is primarily based on the assumption that those who cannot realise benefits from a given PES will simply refuse to participate. Unfortunately, and as is always the case, it is never this simple. The assumption that future benefits are sufficient is a gross oversimplification which must be treated with caution.

Firstly the targeting of the 'poor' resource owners can be difficult as they may not necessarily be the most efficient providers. But beyond this, when a 'poor' supplier is targeted, invisible barriers may prevent his enrolment. Perceptions of any future gain will run alongside attitudes, thoughts of risk, as well as social norms. The decision to adopt a new scheme may be considered more risky than the status quo, no matter how destitute the original option. Moreover, these attributes can vary between the sexes; men and women may carry out very different production activities and have varying stimuli and obstacles but both will inevitably be affected by the application of a PES scheme.

PES schemes have much to learn from past work on engendering development interventions; PES must attempt to understand gender roles, rights and opportunities in its design in order to promote sustainable, equitable and fair interventions. In this ever-changing environment, the physical and social setting of changing climate and modernisation continue to exacerbate gender inequality and inequality more generally. PES schemes must promote equity, reduce marginalisation and promote mechanisms which protect against the threats to the livelihoods of all, including vulnerable women.

This paper focuses on a possible marine PES scheme, and so we must ask: how does this mechanism translate to PES schemes more generally? This said marine PES programmes have much in common with terrestrial ones and we believe the results are relevant to the wider field of PES in general. While social capital and income diversification are important within fishing villages (Allison & Ellis 2001; Visser 2004), they are a significant characteristic in many lower income areas (Barrett 2001; Ellis 2000). As such we believe these results are more broadly applicable.

We conclude by noting that more research on the significance of risk mitigations strategies and safety nets with respect to participation in PES schemes is warranted, as well as more broadly, in other development schemes. And although it is beyond this study to determine the underlying reason for associations between dependency networks and participation, it is important to note the potential affect such networks can have in a female's adoption choice.

Chapter 7

Investigating fishers' preferences for the design of marine Payments for Environmental Services schemes

7.1 Overview

We determine the effects of various management restrictions on adoption rates of marine Payments for Environmental Services schemes. Choice experiments are used in order to determine how fisher participation rates change under different marine PES programme designs. Various designs, with differing restriction rates, show different rates of adoption. However, fishers show a high utility loss associated with any move away from the current management situation, irrespective of restriction levels. This indicates that PES scheme costs may be high and creating an enabling environment could be important to reducing perceived losses, as could investment into conditional in-kind compensation mechanisms. The chapter also shows choice experiments to be a useful tool in marine PES design.

The chapter proceeds as follows. Section 7.2 presents a summary of the importance of appropriate instrument design within the marine conservation setting. A review on fisher preferences for management options is presented in Section 7.3. Section 7.4 presents the study area, after which Section 7.5 introduces the methodological background and the choice model, discusses the use of choice modelling within fisheries management and goes on to describe the choice experiment in more detail. Results are presented in Section 7.6. A discussion of the findings and their policy implications is found in Section 7.7. Conclusions are given in Section 7.8.

7.2 Introduction

In the past decade PES have attracted increasing interest as an innovative conservation instrument. PES seek to address market failures whereby environmental services are not attributed their true value, and increase investment into resource conservation. More specifically, PES attempt to capture those economic benefits derived from environmental services, such as clean water, and channel them back to the ecosystem managers who frequently benefit less from resource conservation than alternative land uses (Engel et al., 2008; Pagiola et al., 2005).

PES are defined as a voluntary agreement between a service provider and a service buyer (Wunder 2005). Inducing participation is central to the success of PES as a policy instrument: potential service providers must voluntarily agree to enrol in any programme design (Newton et al. 2012).

Studies relating to PES participation have increased in the past few years. These have mainly been limited to the study of design factors which improve cost-efficiency (Petheram & Campbell 2010), as well as the implications project design can have on equality across stakeholder participation (e.g. Zilberman et al. 2008). More recently, the literature has looked towards addressing the need to understand potential providers' willingness to participate in PES (Gong et al. 2010; Ma et al. 2010; Newton et al. 2012; Petheram & Campbell 2010; Zbinden & Lee 2005). However, these studies have mostly concentrated on describing endogenous individual and household determinants influencing adoption or non-adoption of PES schemes by service providers. While such information can be useful in targeting households and/or communities for PES interventions, these factors are often inflexible and of limited service to policy makers (Ruto & Garrod 2009).

In practice, very few studies have considered those elements of programme design which induce service provider participation. The influence that design factors exert over a scheme's attractiveness have recently received attention within the context of AES (Ruto & Garrod 2009). AES have much in common with PES in that they are voluntary, incentive-based, conditional and pay for delivery of a desired landscape/land use (Dobbs & Pretty 2008; Ferraro 2008). These recent studies have shown that AES design can indeed influence participation of service sellers. Ruto & Garrod (2009) show that schemes which were designed to be more flexible and offered shorter contracts required lower financial incentives to induce participation. Similarly, Espinosa-Goded et al. (2010) found that those programmes which allowed the maintenance of agricultural activity and did not impose stringent restrictions on farm management were also adopted at lower contract prices. Although not directly relating to AES *per se*, Qin et al. (2011) found that farmers in China were highly concerned with property rights. The provision of priority rights for contract renewal significantly increased farmers' marginal willingness to pay for of existing forestland contracts.

To a greater extent, policy design can be extremely important in achieving adequate acceptance and compliance within the fishery sector and will be particularly important in rural and low-income areas where monitoring and enforcement efforts are often low and/or extremely complex (Christie 2004; Lundquist & Granek 2005; McClanahan et al. 2005). Combined local fishery and conservation goals can be achieved through the

merging of diverse management measures. Closed areas and gear modifications jointly will be needed to address wider scale issues of overfishing (Worm et al., 2009). However compliance, particularly in poor and rural settings, will hinge on community acceptance of any conservation modifications. Previous interventions, principally designed with little consensus from local fishers, have largely failed because they were unable to inspire compliance (Ferse et al., 2010; Pomeroy et al., 2001) or cover the opportunity costs of these low-income communities with few alternatives (Mohammed, 2012). For this reason, understanding how local fishers' value management restrictions is of the utmost importance.

Within this paper we concentrate on how the design of PES instruments can influence participation within a marine setting, a topic which, to date, remains largely unaddressed by the PES literature both terrestrially and within the marine context. This paper uses choice experiments (CE) to investigate some aspects of marine PES design. To date there is little application of CE within fisheries management (Wattage et al., 2011), more specifically, how restriction infrastructures may lower or induce participation by local environmental providers. In doing so this paper highlights the importance of community participation and input at the earliest stages of PES design. CE is also shown as a useful tool in assessing service provider trade-offs, and ultimately for marine management design.

7.2 Fishers and management schemes

Within small-scale artisanal fisheries, marine management has generally favoured regulatory solutions. Of these, the most prolific are MPAs (Agardy et al., 2003). Total prohibition of fishing is ultimately the most effective management option for environmental rehabilitation and conservation; evidence of environmental benefits from regulated MPAs is clear (Agardy, 2000). However, MPAs may not be the most economical, nor the more socially just. MPAs can be inefficient and ineffectual, and can further pose unrealistic and unjustifiable burdens on local low-income fishing communities (Cinner et al. 2009a). In reality, MPA success has been mixed: site-selection can favour less accessible and less degraded areas; resource use often leaks into surrounding areas; and designated areas are often too small in area to protect the wider seascape (Cinner 2010; Graham et al. 2008; Lele et al. 2010).

Restrictions on environmentally damaging fishing gears can form another type of conservation intervention; certain fishing gears have a higher propensity over others to negatively impact the marine environment (Akpalu, 2010). The use of more destructive gear types can: increase physical damage to the substrate; capture a high proportion of juvenile fish; target species important to reef resilience and deter others from fishing

sustainably (Akpalu 2010; Cinner 2010). As such, gear restrictions can be a further effective fisheries management tool and often receive higher support from local fishers (Cinner et al. 2009a). However, the management of artisanal fishers, including the gear they use can be difficult due to their loose, and often poor, organisation (McClanahan & Mangi 2004).

Moving towards more sustainable fisheries often requires a reduction in effort or a switch in methods; both of which pose short-term costs on vulnerable fishers. PES have the potential to complement existing marine management instruments through the provision of short-term incentives. Where local costs are high in the initial stages of restriction measures – whether they be a spatial or gear restriction – PES can assist in compensation for loss of catch, for example. PES should not be viewed as an instrument working in isolation but one that supports current management tools.

Whilst PES may be able to address some of the immediate issues of compensation, they will still need to consider local situations and preferences in order to be successful. Fishers have been documented to hold varying preferences for conservation management restrictions (Cinner et al. 2009a; McClanahan & Mangi 2004). Stakeholder involvement in the early stages of marine conservation development and implementation has been identified as one characteristic of successful approaches (Leslie, 2005; Lundquist and Granek, 2005). Careful consideration of the receptivity of these communities and fishers to design and implementation of conservation interventions is essential for long-term success (Christie, 2004).

Analysis of fisher trade-offs will have numerous benefits. Identification of trade-offs, and resulting design will improve adoption of conservation instrument by local actors. Furthermore, if one assumes that fishers show preferences for the PES design[34] which has the lowest utility cost to them overall, this may lead to more cost-effective PES design.

7.3 Study area: Mtwara region, Tanzania

Tanzania's coastline supports approximately 25% of the country's 43 million strong population of which a high proportion rely on coastal fisheries as a source of food and income. Most marine extraction activities are conducted within the shallow near shore waters (Gustavson et al. 2009; Silva 2006). As population and fisher numbers continue to increase, these coastal resources come under increasing pressure; Tanzanian marine

[34] PES design is considered herein to include various levels of restrictions faced by fishers. This will include facets of MPA restriction such as area under closure as well as further restrictions placed on gear. In reality MPA design will be an integral part of PES design, whereby PES refers to the addition of a compensation mechanism to restricted extraction and/or access.

fisheries have suffered a significant decline in biodiversity and productivity in the past three decades (Silva 2006).

In response to growing concerns about dwindling coastal resources and food security, the Tanzanian Ministry of Natural Resources and Tourism established the Marine Reserves Park Unit (MRPU). The MRPU's mandate is to establish and ensure sustainable conservation of areas of outstanding marine ecological importance, and to manage them in partnership with the coastal communities. Management activities include patrolling of enforced no-take zones and gear restrictions; these are supported through the enactment of village and district by-laws (Silva 2006). However, the MRPU has met some community resistance due to local perceptions of loss, particularly through tighter enforcement of gear restrictions.

The study area located within the Mtwara region of southern Tanzania show a similar pattern to national figures. Within the study area, Malleret (2004) describes a high dependence on marine resources within coastal villages; in some as many as 63-74% of households report high dependence. Fisher numbers also continue to increase: registered fisher numbers within the region have more than doubled since 1996 (Dadi 2010).

7.4 Methodology: choice experiments

7.4.1 The choice experiment

Fishers' preferences for various PES management options were elicited using a CE (Bateman et al. 2002; Louviere et al. 2000).

CE is a survey-based stated preference (SP) technique comprising several choice sets which each contain a set of mutually exclusive hypothetical alternatives. Respondents are asked to choose their preferred option – the one which will give them the highest (anticipated) utility. Each alternative is defined by a set of attributes which take on one or more levels and, as such, the choices are implicit trade-offs between attribute levels (Louviere et al., 2000).

Unlike the more commonly used contingent valuation method (CV), CE enables environmental changes to be described and valued in terms of a specific set of characteristics. Furthermore, with the inclusion of a cost or payment, marginal utility estimates can easily be converted into willingness to pay (WTP) or willingness to accept (WTA) estimates for changes in these attribute levels. In this way, information can be gathered on (a) those attributes which are significant determinants of the 'good'; (b) the relative importance of individual attributes; (c) an individual's marginal rates of

substitution between attributes; and (d) the associated utility cost or benefit of each of the different combinations of attributes (Louviere et al., 2000; Wattage et al., 2005).

SP approaches have received much debate regarding their merits and limitations within the academic literature. Much of this criticism centres on the technique's hypothetical nature. This hypothetical bias arises when people overstate their WTP for a good due to the absence of real economic commitments (Mitchell and Carson, 1993; Neill et al., 1994). This hypothetical bias has been shown to be higher for those respondents who are less knowledgeable, for unfamiliar changes and for voluntary payments vehicles, such as WTA rather than WTP formats (Atkinson & Mourato 2008). In addition, CE have been criticised for increasing the cognitive burden placed on the respondent; the presented attribute-based scenarios may be more complex and there is a limit on the amount of information respondents can meaningfully handle while making a decision. This in turn can give rise to further problems of: learning and fatigue effects leading to apparently irrational choices; increased random errors associated with complexity of task; and satisficing rather than utility-maximising behaviour (Hanley et al. 2001).

As with other SP methods, CE success depends critically on having an accurate, meaningful and understandable scenario; hence careful survey design is essential. The additional information that CE can glean about respondent's preferences has led to many viewing CE as having an advantage over CV. Indeed, over the last decade CE has been increasingly used to value the effects of changes in environmental attributes, and more recently different characteristics of policy design (Hanley et al. 2003; Ruto & Garrod 2009).

7.4.2 CE and fisheries management

To date there has been little application of CE within fisheries management (Wattage et al., 2011). Of notable exception are the works of Wattage et al. (2011; 2005) and Aas et al. (2000). Wattage et al. (2011) uses a CE approach to determine the economic value held by the Irish public for the conservation of deep-sea corals using MPA variant management options. Wattage et al. (2005) demonstrated the applicability of CE in the evaluation of three over-riding management options and its ability to offer meaningful information to the management process. Furthermore, Aas et al. (2000) showed CE to be particularly useful in the evaluation of various fishery management options for harvest regulation within a Norwegian recreational fishery. However, despite a growing use in the industrial fishing arena, CE has been little used within low-income rural settings terrestrially and indeed coastally (Glenk et al., 2006).

7.4.3 CE design

In implementing a CE all recommendations available for SP approaches are relevant to choice model experiments. Questionnaire design followed the principles laid out by Bateman et al. (2002). Surveys collected data on: individual and household demographics; household assets; attitudes relating to fishing, the environment and conservation; and fishing practices, income and livelihood diversification strategies.

The CE revolves around fishers' preferences for various PES management restrictions. After reading a scenario relating to the implementation of a prospective PES programme, respondents were presented with a series of choice sets illustrating possible PES programme options and asked to choose their most preferred. The hypothetical options were presented as possible governmental and marine park authority PES conservation programmes[35]. The following sections describe the key elements of the CE: the selection of attributes and levels of the possible PES scheme, the scenario, the experimental choice card design and the data collection.

7.4.3.1 Attribute and levels selection

The first step in implementing the CE is the determination of realistic attributes and attribute levels to be used within CE scenarios (Bennett & Blamey 2001; Hanley et al. 2001; Mogas et al. 2006).

The selection of relevant attributes and attribute levels was based on information gathered from peer-group meetings and semi-structured interviews, current management options, as well as management options that an implementing organisation was able to influence through policy design. Peer-group meetings and interviews were conducted within each of the six fishing communities chosen for research and were further sub-divided for fisher-types. Appropriate marine management attributes were thus selected based upon importance to fishers as identified in community focus groups and interviews as well as to fit relevant locally applicable management options. In order to minimise issues of cognitive burden, particularly within communities unaccustomed to CE techniques, management scenarios were constrained to the two most relevant attributes: gear restrictions and area closures (i.e. the agreed practices of the PES scheme). Both management measures are considered to be credible and realistic for the areas in question; past governmental interventions have in fact involved net restrictions and marine zone closures. A third attribute relating to the compensation payment package (i.e. the incentive of the PES

[35] Within Tanzanian marine parks fishing rights are controlled by the Marine Park Authority, however outside of marine park boundaries management is in the hands of the Tanzanian Government fisheries division.

scheme) was further included. The payments were described as weekly compensation payments for changes brought on by PES management design. The CE attributes and levels used are displayed in Table 7.1.

Table 7.1 Attribute and attribute levels in choice model experiment

PES scheme attribute	*Description*	*Attribute level*
Size of no-take area	Area as % of current fishing area in which fishing will no longer be permitted and declared MPA.	0*, 10, 25, 50
Size of permitted net meshing	Net mesh size in inches permitted that fishers are permitted to use within fishing grounds. Mesh size is measured as size when mesh pulled at each corner.	1, 3, 6
Compensation payment	Weekly payment in Tanzanian Shillings (TSh) made under PES scheme[36].	-1000, 0, 5000, 10,000, 20,000

Current legal attribute levels, e.g. the status quo, is displayed in red. *Fishing is currently permitted within the marine park (MBREMP) which is officially a multi-use park.

Large differences were noted in the PES management attributes fishers preferred during the piloting stages. At some levels certain restrictions were considered highly beneficial to some fishers while highly detrimental to others, e.g. some fishers preferred smaller net meshing while others favoured larger nets. After piloting it became apparent that some fishers were willing to pay for 'more attractive' management options such as the legislation of small meshed nets. As such an additional negative compensation payment option (-1000 TSh) was included alongside positive compensatory payments (Table 7.1). This would assess if some fishers valued these losses highly enough to be willing to accept negative compensation, e.g. willing to pay for the instatement of 1" nets, which are currently illegal.

7.4.3.2 The scenario

The following hypothetical scenario was presented to fishers:

"I want you to think about the current law and about further prohibitions in your fishing area, more specifically the introduction of additional no-take zones and the prohibition of certain gears. These changes come with compensation for these additional restrictions.

I am going to show you three choice cards. Two cards will show you new fishing regulations and the third card shows you the current regulation in your fishing area.

Each card has two attributes relating to the possible changes in law which can change:

- *The percentage of your current fishing area to be closed to fishing*

[36] Payments reported as US$ equivalent where US$ 1 is equal to 1450 TSh.

- *The allowable net mesh size (in inches)*

The final attribute on these cards is a monetary value. This is the level of compensation per week you would receive if these restrictions were put in place. Please remember, the values shown in BLUE are payments you would receive. Values shown in RED are payments you would make each week to have the new restrictions put in place.

Monitoring and enforcement would be a collaboration between the community and the Marine Park Authority/BMU. Payments would be made monthly and all payments would be withdrawn if the restrictions were not followed.

Please consider carefully which of the scenarios on the cards you prefer, thinking about how each restriction would effect your fishing catch, the compensation you would receive and the trade-offs between the three."

7.4.3.3 Experimental choice card design

A full-factorial design of attribute levels (i.e. 4 levels of no-take area size attribute * 3 levels of mesh size attribute * 4 levels of payment attribute) produced 48 possible PES management scenarios. The management alternatives were then reduced to 16 using orthogonal design, generated using Kocur tables (Kocur et al. 1981). Attributes and attribute levels were piloted. After the first round of pre-testing, compensation payment attribute levels were found to be too low and suitably adjusted. Prior piloted CEs were dropped from subsequent analyses.

In addition to the adjusted attribute levels, it was noted that the combination of small 1" nets with a negative compensation payment (i.e. indicating fishers willing to pay for the implementation of this regulation) were not present within the 16 cards selected via orthogonal design (that is, negative payments were only included for 3" and 6" nets), although this was a management option which many fishers seemed to prefer during piloting as noted above. Therefore, two additional cards were included in the final CE design which combined the negative compensation value (-1000TSh) with small meshing 1" nets (and various degrees of closure: 0 & 10%). A total of 18 cards were therefore used in the final experiment[37].

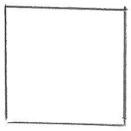
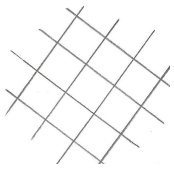
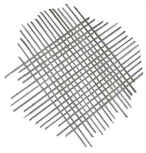
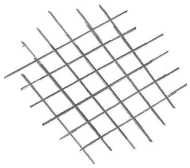
Following an explanation of the hypothetical PES scheme (i.e. the scenario in Section 7.5.3.2), respondents were presented with six choice sets. Each choice set contained three PES options: two (generic) alternatives 1 and 2 and the current status quo

[37] It should be noted that through the addition of two extra cards, the final set of cards presented is not fully orthogonal in design. However, when analysis is undertaken using a logit regression framework, as in the present case, orthogonality although desirable is no longer essential for the method to work satisfactorily. Hence the inclusion of these additional cards should not have a marked impact in the parameter estimates.

baseline (Option 3). Choice cards 1 and 2 were picked at random by the enumerator without replacement from a bag containing all 18 scenario cards.

A choice set example is illustrated in Figure 7.1.

Figure 7.1 Example of choice set

Attributes	**Management Option 1**	**Management Option 2**	**Status Quo**
Closure % closed of current fishing grounds	10 	50 	0 
Net mesh size in "	6 	1 	3 
Payment (TSh)	10,000	5,000	0

The programmes presented possible size of marine area to be designated no-take zones and gear restrictions placed on allowable net sizes (i.e. size of mesh). Monetary compensation was offered as a weekly sum in local currency (Tanzanian Shillings: TSh) but is reported within the results as the US$ equivalent.

Education levels among fishers were found to be low: 96.2% of fishers sampled claimed to have no formal education or attended school only at the primary level. In order to improve respondent's understanding of management scenarios and improve familiarity with possible changes, visual aids were used to represent attributes and attribute levels (Figure 7.1). Visual aids have been shown to reduce task complexity and improve choice by increasing understanding within low-literacy respondents (Jae and DelVecchio, 2004).

Fishers were also run through an example before starting the CE as further explanation and for enumerators to judge fisher comprehension.

7.4.3.4 Data collection

Primary data was collected from six coastal villages located within the Mtwara region of southern Tanzania. Face-to-face interviews were administered with local fishers by trained local enumerators. Between April and July 2010, fisher surveys and the CE were conducted with village fishermen[38].

Initially fishers were targeted using random selection from lists provided by the local village leaders. However, it quickly became obvious that fishers' unpredictability meant a less probabilistic sampling method was necessary. Initial pilot meeting identified some fishers to be sampled; further fishers were selected within villages and landing sites using a non-probabilistic opportunistic sampling method.

7.5 Results

7.5.1 Descriptive results

After exclusion of incomplete questionnaires and initial pilots, the sample size was 317 fishers.

Table 7.2 displays the key demographics for the final sample as broken down for villages and overall. Average fisher age was 35 years and household size was 4.9. Education levels were low across all villages; in all villages fishers having attended secondary school was lower than 7% of the final sample. Table 7.3 indicates the mean fishing characteristics of sample respondents by village, as well as grouped for in and outside of the park. Villages appear to have apparent disparities between fishing and non-fishing income activities across villages. For example, average fishing income was as high as US$ 4.81 a day in Pemba but as low as US$ 1.31 in Mngoji. Furthermore, the number of fishers with other income sources also varied across villages, Mkubiru indicated 71% of fishers claimed non-fishing income revenues; in Pemba village this figure was only 26%. These results could highlight different levels of dependence on fishing as a livelihood.

[38] Women were omitted from CEs. Primarily female fishers were excluded from CE due to the nature of the fisheries in which they participate. Although women would be able to participate within a marine PES with restrictions on net size, gear modification would be unviable for these women and it is likely that selection of scenarios would rely solely on inclusion of small meshed netting; tandilo fishing relies on extremely small meshing to catch 'dagaa' or local sardines.

Table 7.2 Mean demographic characteristics of sample respondents

	In			*Out*			*All*	*In*	*Out*
	Mkub	*Mngj*	*Msim*	*Mkdn*	*Naum*	*Pemb*			
No.	75	39	62	33	58	50	317	176	141
Age	35.5	37.3	32.9	43.0	33.6	33.5	35.3	35.0	35.7
HH_size	4.6	5.4	4.3	5.5	5.5	4.4	4.9	4.7	5.1
Education (% sample)									
None	25.3	18.0	38.7	36.4	27.6	24.0	28.4	28.4	28.4
Primary	69.3	79.5	58.1	63.6	65.5	74.0	67.8	67.6	68.1
Secondary or above	5.3	2.6	3.2	0.0	6.9	2.0	3.8	4.0	3.5

Where: In=villages located in park, Out=villages located outside of park; Mkub=Mkubiru, Mngi=Mngoji, Msim=Msimbati, Mkdn=Mikindani, Naum=Naumbu, Pemb=Pemba.

Table 7.3 Mean fishing and alternative occupation characteristics of sample respondents

	In			*Out*			*All*	*In*	*Out*
	Mkub	*Mngj*	*Msim*	*Mkdn*	*Naum*	*Pemb*			
Fishing income as daily wage:	2.43	1.57	1.31	2.64	1.91	4.81	2.44	1.87	3.10
Weekly fishing income	17.05	11.01	9.20	18.57	13.41	33.79	17.11	13.12	21.77
% with non-fishing income source	0.71	0.51	0.32	0.33	0.41	0.26	0.44	0.53	0.34
Average area of cultivated land	1.97	3.27	2.83	2.64	1.18	0.90	2.05	2.56	1.42
% currently employing illegal gears	0.34	0.10	0.18	0.34	0.05	0.02	0.14	0.19	0.09
% who in past employed illegal gears	0.52	0.82	0.59	0.48	0.71	0.68	0.63	0.61	0.35

Where: In=villages located in park, Out=villages located outside of park; Mkub=Mkubiru, Mngi=Mngoji, Msim=Msimbati, Mkdn=Mikindani, Naum=Naumbu, Pemb=Pemba.

7.5.2 Econometric modelling

The CE approach enables consumer preferences to be modelled in terms of the utility derived from the attributes of a good rather than the overall good *per se*. Statistical analyses of the decision results from (repeated) CE choices can be used to derive the marginal values of a characteristic or the WTP for a more desirable combination of attributes.

Several methods have been suggested for CE estimation. For fractional factorial designs with three or more choices, a multinomial logit model is most commonly used (Heiss 2002). The conditional logit model (CLM) is an appropriate extension of the multinomial logit for those circumstances when the choice between alternatives is modelled as a function of the attributes of the alternative portfolios as well as the characteristics of the respondent making the choice (McFadden, 1974). The CLM estimates the probability that individual *i* chooses alternative *j* as a function of the attributes as they vary between alternatives and unknown parameters as described by McFadden (1974):

$$Pr_{ni} = \frac{e^{x_{ni}\beta}}{\sum_j e^{x_{nj}\beta}} \quad (1)$$

A relatively simple approach, the CLM assumes homogenous preferences across respondents and independence from irrelevant alternatives (IIA[39]). More specifically, the CLM: (1) can represent systematic but not random taste variations (e.g. those that can be linked to observed respondent characteristics but those which cannot be explicitly modelled); (2) displays restrictive substitution patterns (e.g. assumes all pairs of alternatives are equally similar or dissimilar); and (3) is able to model situations where unobservable influences are independent but unable where correlation is generated between alternatives (Hoyos 2010; Hensher et al. 2005).

However, such assumptions frequently do not hold true. In order to accommodate such possible IIA violations within the CE, a nested logit model (NLM) can be utilised. It avoids the need to rely on IIA by modelling choices in a hierarchical nested structure. This device allows error terms across choices within each 'nest' to be correlated with one another, although choices across 'nests' are still assumed to be uncorrelated (Heiss, 2002). Error terms are assumed to follow a Type B Gumbel extreme value distribution as shown in equation (2) (as opposed to the conventional extreme value distribution assumed for the CLM model). The degree of correlation between the error terms is captured by the parameter ρ. Indeed, the CLM can be regarded as a special case of this model when the parameter ρ takes a value of one.

$$f(e_1 \ldots e_J) = \exp\{-[\exp(-\rho^{-1}e_1) + \ldots + \exp(-\rho^{-1}e_J)]\}^{\rho} \quad (2)$$

[39] IIA states that the ratio/likelihood of choosing any two choice options will be unaffected by the attributes or availability of the other options present, that is that the ratio of probabilities of any two options is independent of the choice set (Hausman and McFadden, 1984). Put more simply, all pairs of alternatives are equally similar or dissimilar (Hensher et al. 2005).

In a two tier choice structure, the probability of choosing a particular alternative *k* out of the *n* second stage options, conditional on having selected a particular alternative *j* out of the *m* first stage options, can be expressed as indicated in equation (3). The logarithm of the denominator of this expression is known as the *inclusive value* (*I*), because it summarises the information about the alternatives included in this lower nest. Inserting this inclusive value as an explanatory variable in the first stage of the decision tree yields the expression for the unconditional probability of choosing option *j* out of the *m* first stage options, given in equation (4).

$$P(k \mid j) = \frac{\exp(b_{k|j} X_{ik})}{\sum_{n} \exp(b_n X_{ik})} = \frac{\exp(b_{k|j} Xik)}{\exp I_{k|j}} \tag{3}$$

$$P(j) = \frac{\exp(c_j X_{ij} + \rho I_{k/j})}{\sum_{m} \exp(c_m X_{ij} + \rho I_{k/j})} \tag{4}$$

The model can be estimated by maximising the log-likelihood function is as stated in equation (5), where *y* is an indicator variable which takes a value of one when person *i* chooses option *k* (and thus, by implication, option *j*).

$$\log L = \sum_{i} \sum_{k} y_{ik} \log[P(k \mid j)P(j)] \tag{5}$$

Like the CLM, the NLM only uses information on the first best option identified in each choice set.

Data is analysed in the first instance using the CLM as well using a NLM where appropriate. Models are estimated using STATA 11 software. All variables used within econometric analysis are listed in Table 7.4. Attributes closure and payment entered the models as continuous variables as described in Table 7.4. A large dichotomy was seen in preference for small meshing between fishers so 'Size of permitted net meshing' (Table 7.1) was entered as two dummy variables: 'Net_{small}' where minimum legal meshing was 1" and as 'Net_{large}' where minimum legal meshing was 6", these were contrasted to the baseline of 3" mesh size as this is the current legal status quo.

A modelling constant (here the ASC) is included in the model. The role of the ASC is to account for any unobserved variation in choices that cannot be explained by either the attributes or socioeconomic determinants.

Table 7.4 Variable list and descriptive statistics of independent variables

Variables	*Definition*	*Mean*	*SD*	*min*	*max*
Closure	Continuous variable for % marine area designated no-take zone and closed to fishers relative to current fishing grounds: 0; 10; 25 & 50% closure.	13.5	17.9	0	50.0
Net_{small}	Dummy for net with 1″ mesh size	0.2	0.4	0	1.0
Net_{base}	Dummy for net with 3″ mesh size, current Tanzanian legal mesh size	0.7	0.5	0	1.0
Net_{large}	Dummy for net with 6″ mesh size	0.2	0.4	0	1.0
Payment US	Weekly payment offered as compensation for implementation of new management scenario. Payment transformed into US $: -0.690; 3.448; 6.897; 13.793.	3.7	5.1	-0.69	13.8
ASC	Dummy for Alternative Specific Constant/ choosing of status quo	0.3	0.5	0	1.0
Demographics					
Age	Age of respondent (years)	35.0	12.7	16	82.0
Edu	Count variable for respondent's level of education: 2= attended secondary or above; 1= attended primary; 0 = no education	0.7	0.5	0	2.0
Inc	Continuous variable for respondent's annual income from fishing (US $)	862.7	1,215.4	0	10,925.0
Inpark	Dummy for location: village found inside park borders =1; village located outside =0	0.6	0.5	0	1.0
Illegal	Dummy for those fishing having used illegal fishing methods: 1=fish illegally; 0=fish legally	0.2	0.4	0	1.0
Land	Continuous variable for area of land owned in ha; used as proxy for reliance on fishing whereby those with larger holding are assumed to have lower reliance of fishing	2.1	5.7	0	60.0

7.5.3 Econometric results

317 fishers completed the choice task and accompanying survey. Of these, 221 respondents (70.0%) made at least one choice which was a deviation from the status quo (i.e. alternative A or B in the choice set). 96 fishers chose the status quo in all six choices. Of these 96, 68 respondents perceived the status quo to be their preferred option, the main reasoning being a dislike of any form of marine closure. The remaining 28 respondents (8.8% of the final sample) were considered to be protests and dropped from the final analysis. Protest votes arise when respondents do not state their true preferences which can lead to bias in the final utility estimates. Protests were considered those respondents who selected the status quo in all choice sets, made at least one irrational choice and provided no follow up explanation for choices made.

7.5.3.1 The base model

The main estimation strategy relies on the NLM. While the conditional logit assumes uncorrelated errors, the nested logit specifies the error structure more flexibly and allows some correlation within parent-levels. A log likelihood test indicated the IIA hypothesis could be rejected (p-value 0.088); as such the NLM is favoured over the simpler CLM model.

In order to apply the NLM framework to the present data, it must be possible to maintain that the choice problem can be recast as a hierarchical nested structure. An examination of the choice sets presented suggests an obvious interpretation of this kind, where respondents are hypothesised to choose their preferred management option using a two-stage process. In the first instance, respondents are expected to choose between supporting or not-supporting a new 'improved' management scheme. If a change to the current marine management is chosen, respondents then choose between new management Option 1 and 2. The choice path is illustrated in Figure 7.2. In this way, the NLM assumes greater similarity between the new presented PES management options than between these and the status quo (ASC). In contrast, in a CLM choice path, the first level decision is omitted and respondents choose amongst all three management options equally. Analysis revealed broadly consistent results across both models, with slight adjustments in attribute coefficients.

The base model results (i.e. model containing attributes only) are reported in column 1 of Table 7.5. Column 2 reports the results of the CLM for comparison.

Figure 7.2. Illustration of the nested logit model choice path

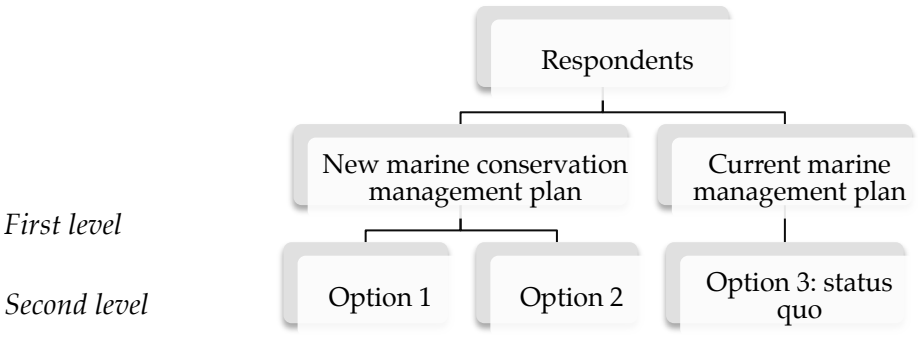


Table 7.5 Model estimates for base specification

	Base model: nested			*Base model: conditional*		
	Coeff		SD	Coeff		SD
Closure	-0.010	***	0.003	-0.013	***	0.003
Net_small	0.075		0.109	0.112		0.127
Net_large	-0.573	***	0.126	-0.700	***	0.119
Payment_US	0.061	***	0.009	0.072	***	0.007
ASC	0.780	***	0.144	0.957	***	0.107
Log-L	-1623.7652			-1625.2184		
Adj-Pseudo R2				0.1308		
Waldchi	62.05			250.69		
Prob >chi	0.0000			0.0000		
N (choices)	5106			5106		
N(cases)	1702					
LR test for IIA P>chi2	0.088					

Robust standard errors have been used. (*) denotes significance at the 10% level, (**) at the 5% level and (***) at the 1% level.

The results reveal that the varying attribute levels influenced willingness to adopt PES schemes. Size of marine closure and having 6" net meshing were negatively associated with willingness to enrol in marine PES (-0.013, p<0.01 and -0.700, p<0,01 respectively). The magnitude of payment offered by the scheme was also a significant determinant and, as expected, showed a positive relationship with willingness to enrol (0.072, p<0.01). The possibility of a PES management scheme which allowed the use of extremely small mesh sizes did not appear to significantly influence fisher's choice. The results indicate fishers show a preference for PES schemes which have smaller no-take areas and that allow the medium mesh size (3"). However, increasing payment associated with PES scheme will enable greater restrictions to be placed upon the conservation area, such as larger no-take zones and mesh sizes. The trade-offs between these attributes are discussed later in the paper.

The ASC was also seen to enter positively and significantly, that is after controlling for all attributes respondents were still more likely to pick the status quo. This indicates a general preference overall for the status quo, and an overall reluctance to engage with management changes.

7.5.3.2 Implicit prices

Inclusion of the payment term within the model enables estimation of the marginal rate of substitution (MRS) between attributes and compensation levels, and indicates the monetary utility loss associated with each management restriction.

Implicit prices are expressed in Table 7.6. As the NLM assumes a linear utility function, implicit prices (IP) are expressed as the ratio of the attribute of interest's coefficient and that of monetary value (Bennett & Blamey 2001).

$$IP = -\frac{\beta_{non\text{-}marketed\ attribute}}{\beta_{monetary\ attribute}} \quad (6)$$

As can be seen from Table 7.6, when all other variables were held constant, closure of an additional 10% of seascape would require an additional US $1.60 a week in compensation. Interestingly, additional net mesh restrictions appear to represent a higher utility cost in comparison. In order to gain acceptance of increased mesh restrictions of 3" to 6" minimum size, weekly compensation of almost US$ 10 per fisher is required; and a 1" increase requires US$ 3.20.

Deviation away from the status quo indicated the highest loss to fishers and indicated an implicit price of US$ 12.72[40].

Table 7.6 Implicit prices: WTA

	Base model: nested	***Base model: conditional***
Closure (US$/10% additional closure)	1.583	1.808
Net_small (3" decrease to 1"net)[41]	-1.222	-1.543
(US$/1" reduction in length of mesh)	-0.611	0.772
Net_large (3" increase to 6" net)	9.351	9.674
(US$/1" additional length of mesh)	3.117	3.225
ASC	12.721	13.239

7.5.3.3 Economic surplus

The economic surplus associated with the implementation of each new alternative management option in contrast to the current status quo can be calculated using equation (7) (Bennett & Blamey 2001)[42].

[40] Calculated from equation (6) where ASC=0.7797918/0.0612984 = 12.72

[41] Deviation to small meshing is also displayed although it should be noted that within the base model this variable was seen to be a non-significant determinant.

[42] The ASC parameter is often ignored in CE welfare measures however conceptually the ASC effect is a component of the indirect utility function and should be included. The ASC can account for unobserved attributes which are known to the individual but not the researcher as well as a 'pure' preference for the current situation (Boxall et al. 2009; Bennett & Blamey 2001).

$$Economic\ surplus = -(1/\beta_{monetary})(V_A - V_0) \quad (7)$$

When:

$V_A = Alternative = \beta_1 Closure_A + \beta_2 Net_A$

$V_0 = StatusQuo = ASC + \beta_1 Closure_0 + \beta_2 Net_0$

Table 7.7 displays the economic surplus of all possible combinations of management strategies associated with the various PES management scenarios.

As expected the greatest utility loss is associated with those management options with the greatest restrictions. Only one management strategy indicated a lower loss, this was via the introduction of smaller meshing and with no closure; however again it should be noted that a deviation from the current 3″ meshing to 1″ was not a significant determinant. Interestingly, fishers perceived restricting net meshing to 6″ would lower their utility slightly more so than a closure as large as half their current fishing grounds, although overall the two were broadly equal in utility loss (a utility loss of 22.1 vs. 20.6).

Table 7.7 Economic surplus under differing management options: US$ -/week

Mesh size (″)	***Size of closure*** (% closure current fishing grounds)			
	0	10	25	50
1	-11.499	-13.082	-15.457	-19.414
3	-	-14.304	-16.678	-20.635
6	-22.072	-23.655	-26.029	-29.987

7.5.3.4 Trade-offs between restriction types

In order to understand any trade-offs being made between the two restriction types, a further analysis was conducted. Trade-offs are calculated using a similar deviation as for implicit pricing whereby the willingness to trade-off between any pairs of attributes is the ratio of these attributes as shown below.

$$Trade\ off = \frac{\beta_{non\text{-}marketed\ attribute\ a}}{\beta_{non\text{-}marketed\ attribute\ b}} \quad (8)$$

Results are presented in Table 7.8. From the data, it appears that fishers approximately equate a twenty percent closure as similar in utility loss to that of a 1″ increase in allowable mesh size from the current 3″ net.

Table 7.8 Trade-offs analysis

	Base model: nested
Closure/Net_large: (10% additional closure)/ 1″ additional length of mesh)	0.508
Net_large/Closure: (1″ additional length of mesh)/ (10% additional closure)	1.969

7.5.3.5 Predicted probabilities: accepting PES design

Predicted rates of adoption are estimated for a number of various PES management scenarios from the base model and displayed in the following tables[43]. Tables 7.9a-c indicate the predicted probabilities of various PES management designs. Table 7.9a and 7.9b display those management designs with only one restriction from the current status quo under the minimum and maximum payment option. Table 7.9c shows the predicted probabilities associated with mixed restrictions under the highest payment.

As can be seen in Tables 7.9a-c uptake of schemes shows high variability dependent upon attribute levels and payments offered. Offering weekly compensation values of 5,000 TSh (US$ 3.5) (Table 7.9a) appeared too low to promote reasonable adoption of the PES schemes investigated; only approximately half of the population would be willing to sign on for the PES design with the lowest restriction of a 10% closure. Raising the weekly compensation payment from US$ 3.5 to US$ 13.8 increased predicted adoption to 70% (Table 7.9b) under this least restrictive scenario.

However, even with such a minimal restriction, 30% of the sample respondents were unwilling to participate. This value rises to approximately 55% for the two harsher restrictions of a 50% closure or a restriction on net mesh size of <6″ independently, even when the highest compensation value was offered (Table 7.9b). One might expect that these relatively low predicted probabilities are due to a high utility cost associated with any move away from the status quo (ASC).

[43] Predicted probabilities are produced using the CLM due to its relative ease of calculation and because results are consistent across both CLM and NLM.

Tables 7.9a-c Acceptance probabilities under differing PES management scenarios.

Table 7.9a Management scenarios with one restriction and minimum payment

Attributes	***PES restrictions***			
	10% closure only/min payment	25% closure only/min payment	50% closure only/min payment	Zero closure/increase to 6" mesh/min payment
Closure (% total area)	10	25	50	0
Mesh Size (")	3	3	3	6
Payment	5,000 TSh (US$ 3.45)	5,000 TSh (US$ 3.45)	5,000 TSh (US$ 3.45)	5,000 TSh (US$ 3.45)
Predicted probability of adoption	53.0	48.1	40.0	38.9

Table 7.9b Management scenarios with one restriction and maximum payment

Attributes	***PES restrictions***			
	10% closure only/max payment	25% closure only/max payment	50% closure only/max payment	Zero closure/increase from 3 to 6" mesh/max payment
Closure (% total area)	10	25	50	0
Mesh Size (")	3	3	3	6
Payment	20,000 TSh (US$ 13.79)	20,000 TSh (US$ 13.79)	20,000 TSh (US$ 13.79)	20,000 TSh (US$ 13.79)
Predicted probability of adoption	70.4	66.2	58.5	56.4

Table 7.9c Management scenarios with joint restrictions and maximum payment

Attributes	***PES restrictions***		
	10% closure/ increase from 3 to 6" mesh/max payment	25% closure/ increase from 3 to 6" mesh/max payment	50% closure/ increase from 3 to 6" mesh/max payment
Closure (% total area)	10	25	50
Mesh Size (")	6	6	6
Payment	20,000 TSh (US$ 13.79)	20,000 TSh (US$ 13.79)	20,000 TSh (US$ 13.79)
Predicted probability of adoption	54.2	49.3	41.2

Again, the predicted probabilities associated with those PES schemes utilising a mixture of restrictions are also low, despite the higher compensation offered (Table 7.9c). Unfortunately the high utility associated with an increase to 6" in net mesh size may override any major trade-off benefits being seen. For example, implementing a 10% closure alongside the 6" mesh restriction reduces the adoption rate by only 2.2%. While this is a good outcome for the implementation of a mixed PES scheme, adoption rate is still very low due to the resistance against increased net restrictions and again the initial move away from the status quo.

7.5.3.6 Robustness check

A selection of socio-demographics variables as described in Table 7.4 were added in an extension to the original model in order to test the robustness of the model findings. Results are shown in Table 7.10. With inclusion of socio-demographic variables, results remain broadly consistent; all significant attributes retain significance albeit to a lesser extent.

Small mesh size 1" (Net_{small}) enters the model as positive and significant at the 10% level, indicating a preference for smaller nets within management scenarios by some fishers. An interaction term between age and Net_{small} (Age_netsm) further suggests that younger men prefer this option. Income interacted with a dummy for the larger 6" nets (Inc_netlg) indicates that higher earners are more likely to prefer PES management scenarios which increase mesh net restrictions to 6".

The ASC drops out as significant once socio-demographics are entered. Income enters as a significant positive determinant of a preference for the status quo and management options which include a movement to larger net meshing.

Table 7.10 Robustness check: model extension with socio-demographic controls

	Base model: nested			*Base model: conditional*		
	Coeff		SD	Coeff		SD
Closure	-0.018	**	0.008	-0.024	**	0.009
Net_{small}	0.611	*	0.359	0.792	*	0.456
Net_{large}	-0.626	*	0.323	-0.774	*	0.406
Payment_US	0.066	***	0.022	0.083	***	0.026
ASC	0.304		0.354	0.563		0.369
Age_close	2.5e-04		1.5e-04	2.7e-04		2.0e-04
Age_netsm	-0.017	**	0.008	-0.022	**	0.010
Age_netlg	0.003		0.007	0.002		0.009
Age_pay	-1.7e-04		4.6e-04	-1.4e-04		5.7e-04
Age_ASC	0.007		0.008	0.005		0.008
Edu_close	-8.6e-04		0.004	3.4e-04		0.005
Edu_netsm	-0.091		0.186	-0.094		0.238
Edu_netlg	-0.271		0.178	-0.317		0.225
Edu_pay	0.005		0.012	0.007		0.014
Edu_ASC	-0.025		0.189	-0.006		0.202
Inc_close	1.5e-06		2.2e-06	9.8e-07		2.6e-06
Inc_netsm	1.2e-04		1.0e-04	1.7e-04		1.3e-04
Inc_netlg	2.3e-04	***	8.6e-05	2.9e-04	***	1.0e-04
Inc_pay	-9.3e-06		7.4e-06	-1.3e-05		8.3e-06
Inc_ASC	2.8e-04	***	8.0e-05	3.0e-04	***	9.0e-05
Log-L	-1531.6516			-1543.9215		
Adj-Pseudo R2				0.1458		
Waldchi	78.89			305.27		
Prob >chi	0.0000			0.0000		
N (choices)	4803			4803		
N (cases)	1637					
LR test for IIA P>chi2	0.0203					

Robust standard errors have been used. (*) denotes significance at the 10% level, (**) at the 5% level and (***) at the 1% level.

7.5.3.7 Independence from previous analysis

The format of the questionnaire was such that the CE followed on from questions relating to the scenario presented in Chapter 6, i.e. the willingness to participate in a marine PES scheme with designated closures and rules relating to gear use. While all attempts were made by enumerators to define these two marine PES participation decisions as distinct it must be noted that fisher decisions could still have been influenced by beliefs of where closures could occur; the CE related only to specific percentage restrictions as yet undefined in space. In order to test if the results were influenced by such predetermined ideas based upon closure location the model was

run again this time including average trust (Avetrust as defined in Chapter 6). Average trust was found to be a positive and significant determinant for men within the previous scenario and so is expected to be so again if fishers are linking the two scenarios. When interacted with each restriction type and ASC variable within the model no Avetrust interaction emerged as significant. We are therefore confident that fishers were treating the two experiments as separate. Results for this analysis can be found in Annex B6.

7.5.3.8 ASC model

Excluding those responses considered protests, the status quo was seen to be the preferred choice in just over half of the choice sets (55.1%). However, 221 respondents deviated away from the status quo (the ASC) in at least one choice set within the CE. This suggests that the status quo was the dominant choice in a number of the sets presented. This is expected as the sets were randomly chosen each time and great variation within fisher's preferences led to few other cards being predominantly chosen.

Table 7.11 ASC model specification

	Base model: nested			***Base model: conditional***		
	Coeff		SD	Coeff		SD
Closure	-0.015	***	0.005	-0.016	***	0.003
Net_small	0.084		0.140	0.093		0.137
Net_large	-0.759	***	0.147	-0.788	***	0.133
Payment_US	0.074	***	0.104	0.077	***	0.008
ASC	0.907	**	0.407	0.946	**	0.389
ASC_inpark	-1.529	***	0.199	-1.535	***	0.198
ASC_illegal	0.489	**	0.198	0.491	**	0.198
ASC_earnings	0.228	*	0.117	0.228	*	0.117
ASC_land	-0.014		0.024	-0.014		0.024
Log-L	-1403.9111			-1403.976		
Adj-Pseudo R2				0.2018		
Waldchi	230.73			258.10		
Prob >chi	0.0000			0.0000		
N (choices)	4803			4803		
N(cases)	1601					
LR test for IIA P>chi2	0.7186					

Robust standard errors have been used. (*) denotes significance at the 10% level, (**) at the 5% level and (***) at the 1% level.

Sixty-eight respondents picked the status quo in all 6 choice sets, 21.5% of the final sample. Given this fairly large selection of the status quo, a further model was run to

determine those characteristics most likely to influence this choice. The ASC model is displayed in Table 7.11. All attributes retain significance within this final model. Coefficients remain fairly consistent in both magnitude and direction. When interacted with the ASC dummy, those who used illegal gear (Illegal) and fishing earnings (earnings) entered the model positively and significantly. Land owned (Land), taken as a proxy for dependence upon fishing whereby larger land holdings allowed further diversification, showed no significant influence on choice of status quo. Location, i.e. those living within the park, (Inpark) was seen as a negative determinant in ASC choice, e.g. those living outside of the park showed a higher reluctance to move away from the status quo.

7.6 Discussion

Design of PES restriction options was seen to influence scheme adoption rates by local fishers. Similar results have been shown in studies in terrestrial PES-like AES (Espinosa-Goded et al. 2010; Ruto & Garrod 2009). Fishers indicated heterogeneous preferences for various marine PES restrictions, indicated by the different utilities associated with the two attributes investigated. Results were comparable across both the NLM and simpler CLM for all regressions.

7.6.1 Trade-offs and participation

As expected, increasing restrictions negatively influenced adoption of PES schemes, and higher compensation payments increased adoption. PES programmes were associated with a high utility loss by fishers; the PES management scenario with the lowest restriction (a closure of 10%) reduced fisher utility by US$ 14.3 per week (Table 7.7): 83.6% of mean weekly earnings. A closure of 25% to current fishing grounds was associated with a slightly higher utility loss of US$ 16.7 a week, almost the average weekly earnings of fishers in the area (US$ 17.1). Furthermore, restricting legal net meshing to a minimum of 6" from 3" had an associated weekly utility loss of US$ 30.0, nearly twice the mean fisher weekly earnings.

Perhaps more interesting than these absolute values are the trade-offs and respective utilities associated with the management restrictions in question. Often marine closures are met with local resistance and gear restrictions can be more readily acceptable (Christie 2004; Cinner et al. 2009a; McClanahan & Mangi 2004). However within the communities surveyed here, it appears that gear restrictions, more specifically the utility loss associated with net restrictions may be met with greater opposition. Fishers equated a restriction of an additional inch on mesh size as approximately similar to a closure of 20%. Accounting for the ASC value, the loss

associated with the prohibition of fishing with meshing less than 6" (weekly compensation of US$ 22.1) was broadly consistent with, if only a little larger, than the compensation associated with a 50% closure (US$ 20.6). However, a 50% closure might appear as a much more extreme intervention from a management perspective.

It should be noted that the net restriction presented herein is a very specific gear restriction, and may have met with such resistance due to local circumstances. Within the Mtwara area, seizure of inappropriate gear is commonplace and carries with it the confiscation of accompanying catch and boat. In recent years, Tanzania implemented a law which outlawed the use of any nets with mesh sizes smaller than the 3" used as a baseline within this study (Dadi 2010). From local focus groups and follow up survey questions, many local fishers felt that even the use of these baseline nets were ineffective at catching adequate fish as overall fish sizes within the coastal areas are small. In addition the most commonly used boat, a non-motorised canoe, did not enable access to the more productive and deeper water areas where fish are larger and more abundant. Indeed, as seen in Table 7.10 higher earners were more likely to prefer those PES interventions which restricted net meshing to 6", perhaps due to the improved ability of larger boats to access deeper waters where larger fish can be caught.

In addition, the lower unit utility losses relating to marine closures could be explained due to a perception that these closures are harder to enforce, hence easier to ignore. Within the area, marine park officials have attempted to monitor possible closed areas with little effect. Moreover, fishers may, quite rightly, believe that their activities can be displaced to new fishing areas outside of the restricted zones, hence decreasing the utility loss associated with this management restriction.

7.6.2 Resistance to change

Another interesting, although perhaps not unexpected, finding was the high utility loss associated with any deviation away from the status quo. When calculating the predicted rates of adoption, increasing the level of attribute restrictions resulted in only a mild decrease in adoption rates compared to the initial PES implementation in the first place. For example, increasing the closure restriction from 10% to 25% was associated with a drop in adoption of only 4.9% when offered 5,000 TSh per week (US$ 3.5) and 4.2% under a weekly compensation package of 20,000 TSh (US$ 13.8). Yet, approximately one third and one half respectively were unlikely to adopt a PES with minimum restrictions in the first instance under the same payment schemes (70.4, Table 7.9b; 53.0, Table 7.9a). Moreover, results indicated that fishers would be willing

to pay as much as US$ 12.7 (74% of fishers' average weekly income) to retain the current management practices, once all attributes had been controlled for[44].

As many as 21.4% of the final sample chose the status quo in all choice sets. Status quo bias is well documented within the CE decision making literature (Boxall et al., 2009; Samuelson and Zeckhauser, 1988). When faced with choices between new alternatives and the status quo, individuals unduly choose the current situation. This decision to remain with the status quo can be motivated by protest beliefs, an inaction to choose, an inability to engage with the more complex experimental design of CE or a genuine preference for the current situation (Meyerhoff and Liebe, 2009). An attempt to limit the incidence of these former three groups was made through the use of a simple and relevant attribute design within the CE. In addition, those respondents who picked the status quo in all six choice sets and did not provide appropriate follow up reasoning were omitted from the final analysis. However, a status quo bias was still noted within the data. Unlike much of the proceeding work in CE and environmental goods, the research herein relates to an initial loss by fishers and not an obvious utility improvement (e.g. loss of fishing grounds and a reduced ability to catch fish), although hopefully with some environmental improvement in the not so distant future. The literature indicates that changes which are considered detrimental (e.g. losses) loom larger on a respondent's mind than any improvements or gains (Kahneman et al., 1991). For this reason, fishers may have shown greater hesitation to participate.

On further analysis it was seen that certain groups were more likely to choose the status quo. Those individuals living outside of the marine park, where current enforcement is weaker and communities have less experience with enforcement bodies, were less likely to choose adoption of an alternative management scenario. In addition, those fishers who had illegal gear (e.g. nets with mesh <3") were more likely to stick with the status quo, even once net attributes had been controlled for. Again, within this sub population, it seems reasonable to expect resistance to change. Illegal fishers are likely to be more dubious of local authorities and the increased restrictions, having had more negative interactions with relevant authorities and perhaps viewing them as less legitimate (Crawford et al. 2004). Fisher perceptions of legitimacy have been shown to be important determinants in compliance behaviour (Hønneland, 2000). Moreover, illegal fishers already function under the base requirements perhaps making adoption of required gear more difficult and costly.

[44] While this is an interesting result it is perhaps worth treating this value with caution. Fishers who experience high poverty are likely unable to pay such a value. However what this value does indicate is a high level of resistance to increased restrictions and interventions within the region.

It was also noted that fishing income was a positive determinant for selection of the status quo. This is an interesting finding. Indeed in many WTP studies, income signifies a budget constraint and is used as a validity test within case studies (Schläpfer 2006; Mitchell & Carson 1993). However, in this circumstance it is a compensation value (a WTA) which is being analysed and income is not a constraint. Indeed, one might expect that those fishers who earn less would be willing to accept less as compensation. Here the selection to remain with the status quo by those who earn more is interesting if perhaps not totally unexpected. Bigger earners, more likely boat owners with high investment into the sector, are likely to be fairly happy with the current perceived situation and reluctant to induce any changes or impose new risks which may impact upon this. Similar findings have been seen with respect to fisher resistance to change practices (e.g. exit a fishery). Pradhan and Leung (2004) found that potential annual fishery earnings was a significant positive determinant in fisher's reluctance to exit fisheries. The same study also indicated those vessel owners who fished using their own boats (e.g. not absentee owners) were more likely to remain. Similar results relating to ownership were seen by Ikiara and Odink (2000). Furthermore, it could simply be a case that the weekly compensation rates offered within the CE were simply too low for higher earner to make adoption worthwhile. Furthermore, when socio-demographic variables were entered into the model the dummy for retention of the status quo was no longer seen to be significant. An interaction term between fishing income and the ASC was seen to be a strong significant positive determinant of status quo choice. This provides further support that those higher earners were more likely to stick with the status quo.

7.6.3 Implications for marine PES

Perhaps two of the more interesting findings are as follows. Firstly, although various attribute levels influence management adoption, hence acceptance, it is possible that within those coastal areas creating an environment whereby change is not met with apprehension and hostility could be equally as important, if not more so. Deviation away from the status quo carried with it a high initial utility cost, comparable and greater than those associated with the restrictions themselves. In such cases, efforts to support local communities, build trust and ease transition to new management practices may be more fruitful and cost-effective, if albeit a little more time consuming at the on-set.

Secondly, overall the cost of a PES scheme may be too high. The hypothetical PES scheme which offered the lowest compensation of US$ 3.5 per week to fishers for a restriction of 10% closure is estimated to be adopted by only 50% of the target population. Moreover, a PES offering a much higher weekly compensation of 20,000

TSh or US$ 13.8 (over ¾ the value of fishers' average weekly income) for the same minimal restriction failed to entice as much as a third of the population. While this may not seem like much, it must be noted that compensation is based on a weekly payment and must be aggregated for an entire fishing community.

This said, the CE methodology can provide information on the most cost-effective intervention for marine PES. For example, herein the perceived utility losses associated with different restrictions are expressed. However, the value of these restrictions for environmental service generation will vary. For example, a closure may require a lower payment but give equal effectiveness in terms of environmental service delivery, or vice versa. Where the science is in place, such methodologies can provide valuable and complementary information.

Furthermore, results indicated that income is a positive determinant for opting out of PES management change. If weekly compensation rates cannot entice higher earners, who undoubtedly are often the highest extractors of the resource, PES schemes are unlikely to accomplish conservation goals (Engel et al. 2008; Wunder 2007). Indeed, within coastal communities fishing incomes can vary widely with some fishers barely catching enough for subsistence, let alone commercial activities, while other can be considered well off by local standards. Payments may be required to reflect all of these population groups, perhaps via differentiated payments. However, differentiated payments bring with them increased opportunity costs and can induce conflict between parties (Jack et al., 2008). Alternatively non-cash incentive structure could be structured and introduced to induce participation. For example, access to storage facilities may enable fishers to better negotiate prices and would increase profits relatively for all fishers involved, so long as access is not monopolised.

7.6.4 Limitations and future research

In order to reduce the cognitive burden associated with CE, design was limited to two attributes, closure and allowable mesh size, with four and three levels respectively. However, this design limited the ability to report on trade-offs and design of appropriate restriction levels. For example, the restriction on small meshing was seen as insignificant. Therefore for gear management restriction was limited to only current and large meshing and limited the management scenarios available.

In addition, that utility loss from a 50% closure of current fishing areas equated to that of an 3" increase in mesh size may generate concern that respondents were unaware of what they were being asked. However, as previously mentioned, it is not unreasonable that fishers might value these smaller meshed nets so highly given local circumstances.

Despite these limitations, the findings herein could be the valuable subject of on-going investigation. Future studies may aim to move beyond this case study and replicate research. In addition, there is scope for more detailed work on those further attributes fishers may respond to, in terms of both restrictive strategies as well as what non-monetary incentives that may induce participation e.g. access to improved markets and storage facilities to name a few.

It will also be useful to identify if those attributes identified herein, as well as additional attributes so far not addressed, continue to be significant determinants over a wider sample of artisanal fisher communities. What similarities lie within case studies as well as those site-specific qualities?

Given the large utility loss associated with a movement away from the status quo, it would also be informative to identify whether this is a common feature within fishing communities. Indeed, as previously noted, reluctance to exit fisheries by fishers has been identified within recent studies (Cinner et al. 2009b; Ikiara & Odink 2000; Pradhan & Leung 2004; Teh et al. 2008). This inertia to change may also transcend into less extreme novel management strategies. On the other hand, the relatively large utility loss recorded herein could relate to site conditions; at least in part, local conditions are anticipated to have played some role in the magnitude of this perceived loss. For example, those communities located outside of the marine park were more likely to stick with the status quo, perhaps due to a greater mistrust of or a reluctance to engage with new and less known regulating bodies. Further studies should identify those circumstances which have culminated to produce this effect as well as those fishers more likely to perceive a loss, as well as those PES interventions which will mitigate this loss.

7.7 Conclusions

Overall the study finds CE to be a useful policy tool in identifying fishers' preferences for various management options. CE enables explicit analysis of trade-offs, as well as and their appropriate levels. CE can assist in evaluating which management alternatives may be of least-cost as well as locally accepted and effective in their conservation goal. This will be key in the concurrent design of appropriate conservation and development tools and in particular cost-effective PES. The CE methodology can also identify those groups less willing to engage in such novel schemes, as well as identifying those aspects of instrument design which may disincentivise participation; in doing so CE can help recognise whether the restrictions are inappropriate if there is a reluctance for change overall.

The research shows that fishers are currently reluctant to move away from the status quo, and that associated costs in promoting this transition will be high. Mechanisms which reduce this initial transition cost are called for, as are conditional non-monetary incentives which can allow fishers to sustain their welfare at a lower cost.

Chapter 8

Synthesis and Conclusions

8.1 Purpose of thesis

The purpose of this thesis is to learn more about the feasibility of the PES instrument within a marine and coastal context, with a particular interest in if and how these mechanisms would translate to the small-scale artisanal fisheries.

The research focuses on both sides of a PES market: supply and demand. In the first part of the thesis we investigate the main challenges and barriers to marine PES programmes, and those issues which need to be addresses in order to improve buyer confidence in the instrument. Opportunities for marine PES to improve current environmental and social performance are explored.

In the second part of this thesis we identify determinants which may promote or discourage fisher participation within marine PES schemes. The thesis concentrates on identifying those determinants, such as social capital and gender, which may be important in low-income households and vulnerable groups. In addition, the thesis investigates the role that instrument design can play in the rate of adoption of marine PES by fishers. Results are based on primary data collected from our case study, a survey of six artisanal fishing villages from the Mtwara Region in southern Tanzania.

The thesis is the first to address the feasibility of PES in the marine and coastal environment, and more specifically within small-scale artisanal fisheries. It is the first to tackle issues of participation, revealing preferences for design and barriers to participation, as well as more overarching issues in implementation.

The thesis uses both qualitative and quantitative data to answer these questions, and is one of the first to use CE within small-scale fisheries and expert elicitation within the field of PES.

8.2 Synthesis of findings

The past decade has seen a flurry of interest in PES from a wide audience; these include scholars, policy makers and conservation and development practitioners alike (Tacconi 2012; Van Hecken & Bastiaensen 2010) and more recently, interest from the marine conservation community (Lau 2012; Mohammed 2012; Murray et al. 2011). The common perception is that PES can address some of the failings of more indirect

instruments, such as ICDPs and CBM. The premise that better off service buyers compensate environmental service suppliers touts dual benefits: a source of sustainable finance for conservation (Ferraro & Kiss 2002) and the ability to promote development needs alongside conservation (Wunder & Albán 2008). However, few PES programmes have been carefully documented and empirical results are scant (Engel et al. 2008).

Moreover, how such schemes will translate to the marine environment remains to be seen and is confined to discussion and policy pieces (e.g. Lau 2012; Mohammed 2012; Pagiola 2008). Artisanal fishing communities represent some of the world's most vulnerable socio-economic groups (Béné et al. 2010), and it is debatable what implications PES can have within these fragile coastal and marine environments.

8.2.1 Examining the issues in marine Payments for Environmental Services through expert elicitation

While previous chapters have dealt with supply-side dynamics, the final chapter in this thesis broadens the narrative to include additional issues associated with implementation, in particular securing demand. In the final research chapter, expert elicitation identified a number of opportunities and challenges for marine PES programmes.

8.2.4.1 Demand

Expert elicitation in Chapter 7 highlighted that demand remains a challenge for coastal and marine PES schemes, as it does more generally. Demand within marine PES is subject to many of the same issues experienced by terrestrial PES, largely relying on evidence of adequate environmental service delivery. Indeed, if marine PES wish to move away from government and donor schemes, dealing with these concerns is paramount.

8.2.4.2 Cementing property rights

Tenure, or the lack of it, was the most prominently cited barrier by experts. With ill-defined property rights the norm in coastal and marine ecosystems, it is hardly surprising that issues of tenure feature as such a key concern. Multiple stakeholders and lack of seascape rights are a common feature of coastal areas. These factors complicate marine management globally, make effective conservation difficult and further cement perceptions of the marine environment as a problematic management issue. In actual fact, realising effective property rights within the marine context could have beneficial effects for marine conservation. Indeed, privatisation has been shown

to have beneficial effects in industrial fisheries (Worm et al. 2009). Although these exact mechanisms are not directly transferable to smaller-scale operations, PES could assist a similar development in artisanal fishing areas. For example, marine PES can work alongside and promote the implementation of TURFs. Programmes which assist local communities to protect and enforce their local areas can inspire improved local governance and promote sustainable practices, removing those incentives to overfish associated with open-access conditions. Indeed, FAO (2008) state that the most critical reform in fisheries policy is the institution of secure marine property rights systems. Reforms will require investment in collective action, strengthened civil society and the empowerment of poorer fishing communities (FAO 2008). Perhaps in response, recent years have seen the generation of a number of new initiatives which transfer property rights to local communities (Lau 2012). Marine PES can promote additional interest and investment in order improve tenure for artisanal communities. As such, perhaps tenure should not be seen as a limitation. Instead PES should be seen as an opportunity to promote interest in tenure, and assist in dealing with a long serving and highly detrimental problem in the marine environment.

8.2.4.3 Monitoring of 'invisible' services

The nature of marine PES and the consequential lack of scientific understanding surrounding these environmental services continue to be strong concerns. Inspiring confidence in buyers may critically hinge on better comprehension of these issues and the environmental services themselves.

Certain low-hanging fruit exist; environmental services with those more simple pathways, such as storm line protection and carbon sequestration, can promote investment into the sector. With a growing number of successful marine PES, will come a confidence which will enable experimentation with more adventurous marine PES. Furthermore, the last few years have seen an improved understanding of how certain marine management decisions impact on ecosystem functioning and significant advances in the understanding of marine ecosystems and consequent valuation of resources (Lau 2012; Barbier 2010; Barbier et al. 2008).

Advances in marine ecosystem science and modelling will only serve to improve monitoring and assist in proving additionality. However, in the mean time marine PES can learn a lot from terrestrial systems where action-based payments are widespread (Wunder et al. 2008). Although marine ecosystems suffer amplified issues of cause and effect, sufficient understanding exists of how many management decisions can increase service flows (Lau 2012).

8.2.4.4 Enforcement

Complex and difficult enforcement relates back to issues of tenure and to a greater degree multiple stakeholders. Issues of enforcement are not limited to marine PES. In actual fact, for most marine interventions enforcement is both costly and complex (Petter Johnsen & Eliasen 2011; Pomeroy 2001).

Inspiring compliance is undoubtedly one of the more important attributes an instrument can do to lower enforcement costs as well as secure long-term success. If PES schemes are to work in coastal areas, they must induce compliance of some of the world's poorest. In rural low-income areas, resource users make their decisions with strong bias towards the short-term. Previous interventions have suffered because costs are immediate whereas the benefits of conserving tend to occur over longer time frames, are indirect and diffuse (Kiss 2004).

Marine PES programmes enable communities to gain tangible benefits on a more immediate timescale. In this way PES may be viewed as more legitimate than previous interventions (if of course contracts are executed with adequate consideration of local circumstance). In addition PES schemes which can assist in securing tenure, as discussed in Section 8.2.4.2, can promote local support and enable local enforcement. Providing a platform which enables communities to participate in enforcement has been shown to significantly improve marine management. At the local level community-based enforcement can be effective in significantly reducing unsustainable practices and can improve compliance (Crawford et al. 2004). Sommerville et al. (2010a) showed that fear of local institutions reduced the likely uptake of previous unsustainable practices. Moreover, moral obligation and social influences can play an important role in improving compliance behaviour (Kuperan & Sutinen 1998).

However, with migrant fishers commonplace, enforcement will need to extend beyond promoting local compliance. Community-based enforcement will be less able to deal with the non-compliance of fishers from neighbouring villages, and poorly equipped to handle illegal fishers from further off. Crawford et al. (2004) show semi-formal village-to-village interactions though village heads can be an effective strategy in dealing with non-complying fishers from neighbouring areas. However, they go on to state the need for more centralised and formal enforcement institutions in dealing with fishers from further afield or where village-to-village mediation may fail.

As such marine PES will require additional institutional and governmental support. Indeed, this has always been an issue for marine management initiatives based in low-income countries. If the PES mechanism can inspire both local and regional support,

and provide additional funds to realise conservation, it could be a step in the right direction.

8.4.4.5 Community contracts and equitable sharing

Given the results above, perhaps most pertinent is how to identify appropriate stakeholders. Marine PES programmes within artisanal fishing villages will undoubtedly rely on community contracts. With this there will come greater complications in dealing with 'poor' stakeholders. Community contracts remove the notion of voluntary participation and will require a complex set of incentives (Sommerville et al. 2009). This relates back to the previously criticised assumption of PES as a voluntary mechanism in which people will simply refuse to participate where benefits are insufficient. In reality there will ultimately be winners and losers. As such, the problem is to what extent should PES and community contracts attempt to deal with local poverty.

Experts showed mixed opinions in the degree to which marine PES schemes should be pro-poor. Concerns related mainly to the loss of environmental efficiency and that in fact PES would be unable to deal with the underlying issues of poverty in the first instance. Both are valid concerns. However, what is evident is that PES should not fortify pre-existing inequalities or further marginalise impoverished groups, something which is all together too easy (Milne & Adams 2012) and perhaps more so within small-scale fishing communities. Marginalisation, exclusion and discrimination are common within these communities (Béné 2003) and, as seen in the previous chapters of this thesis, further barriers exist for vulnerable socio-economic groups. For example, if contracts devolve tenure, females may be at particular risk of being ejected from previous roles, as has been seen in the past (Carney 1993). Moreover, safety nets for non-participants should also be acknowledged in design – particularly in those circumstances where PES hope to promote and cement local tenure and enforcement, possibly changing current social norms.

Again, as previously discussed in Section 8.2.3, the case can be made for non-monetary incentive schemes which can allow widespread of benefits, as well as proportional benefits (e.g. storage and voucher facilities).

Again low-hanging fruit exist within the marine portfolio, for example as previously mentioned: mangrove forests and carbon sequestration. With relatively simple environmental service pathways and a static nature, lessons can be learnt about contract implementation, as well as the underlying distributional implications within coastal communities.

8.4.4.6 Covering initial opportunity costs

Committing to the long-term financing of marine PES may prove to be expensive for conservation organisations and some question whether marine PES can work at the scale required. Indeed, more generally debate still surrounds adequate protection size for marine areas within the wider conservation literature (Claudet et al. 2008; Halpern 2003; Walters 2000).

However, the fishery benefits of MPAs are well documented (e.g. Agardy 2000; Day 2002; Gell & Roberts 2003; McCook et al. 2010 and Roberts et al. 2001). McClanahan (2010) further show how small to moderate fishery closures and gear restrictions can increase fisheries profitability in adjacent areas. However, initial costs exist for fishers and when incomes are directly affected by restrictions, local support for conservation initiatives has been shown to be lower (McClanahan et al. 2009). As such, marine PES may be able to assist with these initial opportunity costs experienced by fishers and thereby mediate conflict.

8.2.2 Determinants of fishers' willingness to adopt a marine Payments for Environmental Service scheme

The success of PES in promoting concurrent objectives of environmental stewardship and poverty alleviation is questionable, and empirical research lags well behind policy design and implementation (Pattanayak et al. 2010). What is clear is that without participation of poor stakeholders, PES holds little promise in meeting these goals.

Results as to the accessibility of PES schemes in the terrestrial setting are mixed. While some found good enrolment by 'poorer' households (Pagiola et al. 2008), other report difficulties for smaller and less educated households (Grieg-Gran et al. 2005; Southgate et al. 2010; Wunder 2008). Issues relating to the lack of necessary skills, labour or capital as well as compromised food security have been cited as concerns (Grieg-Gran et al. 2005; Pagiola et al. 2008; Southgate et al. 2010;Wunder 2008).

The lion-share of PES literature as it relates to participation discusses how one can improve eligibility through project design (Pagiola et al. 2005, 2008; Wunder 2008; Zilberman et al. 2008). Implicit in this research is the notion that if eligible 'poor' service providers will participate, and simply refuse to participate or withdraw where benefits are not realised (Pagiola et al. 2005). This is based on two distinct assumptions: that all potential actors are willing to participate and that PES programmes are indeed voluntary. Both are, however, open to criticism.

First we look more closely at the first of these two assumptions. As demonstrated within Chapters 6 of this thesis, although eligible and able, willingness to participate may be influenced by a number of other factors. Indeed, factors that have been proven to be important in lower-income households (Dercon 2002) can influence an individual's willingness to participate. Beyond household income, ability to spread risk can factor into future decision choices (Alderman & Paxson 1992).

Social capital has been shown as important in fisher livelihoods; Visser (2004) identified fishers endowed with stronger social networks and better financial resources as better able to defend their interests against other fishers. Within Chapter 6 of this thesis we demonstrate that social capital and income-diversification strategies can be significant determinants in the decision to adopt marine PES schemes. Not only this, but various facets of social capital have different implications in participation choice. Indeed, within this thesis we show that levels of overall trust and membership within a non-fishing group significantly increased willingness to participate within a proposed marine PES scheme. Interestingly, quite the opposite was seen for presence within a reciprocal fishing dependency network. More specifically, being dependent on another and/or being depended on for fishing activities, and the number of these networks, reduced the likelihood of adopting the proposed marine PES programme. Yet to date there is little mention of social capital and insurance mechanisms within the PES poverty literature. That social capital can influence decisions to try novel development initiatives is previously undocumented and has interesting implications for PES schemes hoping to target poor households; in particular if poorer households rely on informal insurance networks and are fearful to upset these.

In addition, these determinants may vary by gender. Women have long been recognised as some of the more vulnerable societal members (Harrison 2000), and perhaps to an even greater extent within low-income fishing communities (Porter & Mbezi 2010). However, women continue to be overlooked in development assistance policy design within the fishery sector (Sze Choo et al. 2008), mainly due to the 'invisibility' of their role (Sze Choo et al. 2008; Weeratunge et al. 2010). However, women can also be some of the most environmentally detrimental. Within our study site, tandilo fishing is the sole practice of women. Conducted within the intertidal zone, large mosquito nets are dragged along the substrate by three to six women catching small fry and, in addition, many juvenile fish. However destructive, such fishing represents one of the only sources of income, and indeed independence, for these women.

What is apparent is that in order not to further exacerbate fisher women's vulnerability, marine PES must take note of their role in both environmental impacts and instrument design. As seen in Chapter 6, women demonstrated different motivators for participation. Men were more readily influenced by factors such as previous investment into the fishing sector. Owning boats and working on the larger boats (dhows) increased male resistance to adopt; on the other hand men, with larger farming plots were more willing to sign up to the PES scheme. However, those women more ready to participate showed higher levels of experience overall, for example: education, experience with alternative income activities and working in groups. This is perhaps in part related to the importance of prior experience and education in building confidence.

Social capital was again seen to be an important determinant, although various facets affected the genders differently. The only social capital variable which was seen to significantly influence men was trust; men with more pronounced trust overall were more likely to participate in the proposed marine PES. However, women showed a more profound reluctance to upset pre-existing networks. The variable 'presence within a reciprocal fishing dependency network' was again seen to be significantly deter participation.

These differences are important to note; PES schemes which determine fishers' participation but focus mostly on male determinants – who usually comprise the larger share of the fisher numbers – can as a result exclude female stakeholders. In doing so PES schemes can have two important consequences: the further marginalisation of women within fishing communities and an inadequate environmental targeting.

8.2.3 Fishers' preferences for the design of marine Payments for Environmental Service schemes

In addition to understanding those fisher characteristics which can cause reluctance to sign up to marine PES schemes, these schemes must inspire participation in the first instance. Previous marine conservation interventions, principally designed with little collaboration or consensus from local fishers, have largely failed because they were unable to inspire compliance (Ferse et al. 2010; Pomeroy et al. 2001).

PES programme design can be an important factor in a participation decision choice. Previous work by Qin et al. (2011), Espinosa-Goded et al. (2010) and Ruto and Garrod (2009) show programme design can affect adoption rate of conservation schemes. In addition, McClanahan and Mangi (2004) demonstrated heterogeneous preferences for marine management restrictions. It follows then, that PES programme design, in

particular those restrictions implemented, will have implications for PES participation rates.

In the first instance, Chapter 7 demonstrates CE to be a useful aid in PES design. In the second, the attributes of PES management design were indeed shown to have implications for PES adoption by fishers; as anticipated increasing restrictions reduced the desire to participate. Although hard to compare interventions with differing units of change, fishers approximately equated the loss of half of current fishing areas with a restriction on nets with mesh size below 6" (an increase from the current 3" status quo). Indeed, one might think this a large utility loss to associate with a net mesh increase; 50% of fishing grounds is unlikely to be considered punitive in anyone's eyes. However, fishing by many is limited to those in-shore areas accessible via small, non-motorised one or two man canoes. As it is local fishers complain about the inability to catch fish using the legal 3" meshed nets, and illegal nets with smaller meshing are not uncommon in the area. Increasing to large mesh sizing will only serve to further reduce catch in the in-shore areas.

PES implementation showed high utility loss to fishers overall. However perhaps more interesting, fishers associated the majority of this cost with the initial move away from the status quo vs. the additional marginal restrictions themselves. For whatever reasons, fishers were reluctant to engage with any additional restrictions in the first place and as such showed a strong reluctance to participate in a marine PES scheme. Therefore, while programme design is significant, perhaps more important in the first instance is building trust. Creating a facilitating environment in fact can be more important, and indeed reduce PES compliance costs overall.

Given potentially high compensation values, increased transactions costs in dealing with multiple heterogeneous groups (Jack et al. 2008) and recent evidence that resource management decisions in artisanal fishing communities are not always market-driven (Daw 2008; Pet-Soede et al. 2001), payments need not necessarily be cash. Positive in-kind incentives may lower programme costs overall and decrease conflicts if seen as more legitimate. Incentives which allow fishers to improve negotiating power, for example when selling fish on, will increase all fishers' income proportionally. Incentives could include improved access to storage facilities, a common problem within rural artisanal fishing communities. However conditionality, which is fundamental to PES, must be retained. Incentives could also be linked to improved savings and loans, as has recently been trailed by Net-Works[45]. Net-works is a recent

[45] For more information on Net-Works please visit:

innovation by global carpet tile manufacturer Interface, partnered with Zoological Society of London, to establish a community-based supply chain for discarded fishing nets found along the shore. Payments for nets are made through a voucher scheme which participants can then use to 'buy' savings shares within a local Village Savings and Loans Association. While 'payments' are not specifically made for an ecosystem service – nets are used as a recycled source of nylon for carpets – environmental and development benefits are clear concurrent objectives. Much can be learnt from the initiative about implementation in an artisanal setting for both businesses and PES schemes alike.

8.3 Limitations and further research

This thesis provides insight into some of the challenges and opportunities for PES schemes in small-scale fishing communities. At the same time it raises further questions which could not be addressed within the context of this thesis.

A lack of academic literature pertaining to marine PES meant the Chapter 3 in this thesis relied solely on expert elicitation. While this technique has the advantage of making new knowledge available and highlighting consensus as well as diversity in opinion (González et al. 2007), it suffers a number of limitations (Richards 1996). Without many real world examples for experts to base their knowledge, expertise is drawn from related areas. In addition, where experts refuse to interview, it may not always be possible to obtain a representative sample (Richards 1996). This said, a good spread of expertise was obtained in the final data chapter, thus we hope a reasonably full picture of the state of knowledge in the field. Furthermore, the use of open-ended questions was used in order not to lead the experts' thought process. However, the coding of open-ended questions can be subjective. In order to minimise these issues, coding was iterative and carried out by the author only. Moreover, within the on-line questionnaire, respondents were not motivated to put down all those issues which came to them, and perhaps only cited the most pertinent. As such, it is important to remember that the elements coded within this final data chapter are perhaps only the most relevant in a longer list.

In spite of these issues, much can be learnt regarding marine PES from expert elicitation as has been discovered in this thesis. Given that experts draw on their own knowledge base, it would be interesting to dissect results further and identify if certain issues are more cited by specific fields. Unfortunately this was not possible given the

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http://www.interfaceflor.co.uk/web/about_us/media_centre_landing_page/press_releases/press-On-World-Oceans-Day-Interface-and-ZSL-announce-partnership-to-tackle-environmental-problem-blighting-developing-world-shores

wide range of elements coded within the open-ended questions relative to the final sample size, and would rely on more targeted close-ended questions.

Unfortunately, though unavoidable, the hypothetical nature of the marine PES scheme raises a number of limitations which should be noted. The hypothetical and cross-sectional nature of the data does not allow analysis of those actors who may initially take up a scheme but drop out later, or more interestingly if members may enrol at a later date when the scheme is more well-established and perhaps less 'risky'; nor does it allow examination of other more dynamic aspects of participation, such as implications of changing income for example. Further research, over a time-series as well as focusing on real marine PES schemes can deal with these issues. Furthermore, it is possible that results suffer from hypothetical bias, where respondents have strategically answered in order to frame outcomes. This is not unheard of within those techniques which rely on hypothetical scenarios such as within a stated preference methodology (Chapter 5) (Mitchell & Carson 1993). However, issues can be limited by using accurate, meaningful and understandable scenarios (Hanley et al. 1998).

Primary data from the case study described relied on non-probabilistic sampling methods. Non-probabilistic sampling runs the risk of being biased and failing to be representative of the target population (Dixon & Leach 1982; Kitchenham & Pfleeger 2002). Probabilistic sampling methods are the preferred method. Initially fishers were randomly selected from village lists. However, the unpredictable nature of fishers and the low likelihood of appointments being kept made data collection time-consuming, costly and difficult. In any case, non-probabilistic sampling can be justified for pilot and exploratory work and where a certain amount of knowledge is known about the target population (Dixon & Leach 1982). Efforts were made to target various fishing types and wealth groups. The sample from the case site shows similar characteristics to previous data on fisher communities collected within the area (Malleret 2004; Malleret & Simbua 2004). Furthermore, non-probabilistic sampling methods can suffer from issues of self-selection. One might expect that those fishers opposed to the proposed PES scheme would refuse to partake in the questionnaire and analysis; however, overall the collected sample indicated a large proportion of respondents sampled refusing to participate in the PES scheme.

Despite these limitations, this thesis sheds light on possible barriers to participation and calls for a greater understanding of the effect of risk mitigating strategies and informal insurance on the adoption of marine PES schemes, as well as PES schemes more generally. With the growth of marine PES initiatives, there needs to be an increased understanding of those factors (beyond income) which are important for the

participation of more vulnerable households. Future research should aim to identify further important networks in participating communities, not only with respect to how these might influence participation but how a PES scheme might affect these over the longer term. Indeed, while such networks may prevent participation, PES with its accompanying incentive package may lead to the breakdown of said informal insurance systems. In addition, where marine PES schemes hope to target the poorer members of society, it will be important to determine those elements of scheme design which are preferred and indeed less costly. Within this thesis a limited CE attribute design was utilised to determine if CE was a suitable aid in PES programme design. PES design attributes were limited to two characteristics and payment size in order to reduce cognitive burden. Fishers showed a good comprehension of the CE approach. Future studies should concentrate on including all relevant attributes possible. Although not delved into in this thesis, it would be interesting to break up perceived costs of alternative restrictions by wealth group, and identify proportional costs to various groups.

This thesis is based on one-case study site. It would be useful to see how the results herein transfer across to other marine areas, seeing what similarities and differences are found. Further analysis to determine the extent to which similarities can be found in the terrestrial setting and lessons transferred is advised, and given the more established nature of these PES schemes this may indeed be the place to start.

8.4 Policy Implications

What appears evident is that marine PES schemes have great potential within the coastal and marine environment, and indeed it seems unlikely that the interest in them will wain. Marine PES programmes are perhaps used best to compliment some of the failings of current instruments, for example to cover initial losses in opportunity costs where communities are unable to bear these costs. That PES can inject positive incentives alongside conditionality into current tools is perhaps its greatest selling point; conditionality will likely improve compliance. However, this novel advantage of PES also comes with a caveat: consequences can arise from the addition of this incentive-based component. The implications of which should not be overlooked within fishing communities. This thesis demonstrates that while marine PES can bring opportunities for these communities to secure and sustainably manage their resources, inequalities between households can prevent participation of some of the more vulnerable.

PES schemes which aim to be ‘pro-poor’ may utilise quota schemes which include a certain quota of low income households; however, care is needed when utilising quota

schemes which only look at income. Poverty is multifaceted and extends beyond a lack of income. Access to new livelihoods can be determined by gender, class, ethnicity as well as social norms and customs to name a few (Allison & Ellis 2001). PES schemes which hope to target more vulnerable members of society will need to move away from the assumption that communities can be treated as homogeneous units. This over-simplification ignores that people can be embedded in dependencies and hierarchies, hold different values and therefore respond to incentives and policies differently (Coulthard 2012).

Surprisingly, whilst the type and level of restrictions had a significant impact on the likelihood of marine PES adoption, these costs were negligible when compared to the initial utility loss associated with a move away from current management practice. Fishers showed inertia to move away from current management. As such, marine PES should not assume that just because it offers a 'greater' incentive package over previous conservation models it will simply establish itself and be immediately accepted. Factors other than financial needs have been shown to influence fisher behaviour (Cinner et al. 2009; Daw 2008). In fact, relics of previous imposed and likely failed instruments have left behind legacies of distrust and conflict. As such, it is likely that many marine PES will need to be accompanied by conflict resolution instruments which can reduce this initial cost. Despite this, CE showed itself to be a useful tool in designing and determining possible attributes of a marine PES scheme.

Demand outside of conservation and development NGOs continues to prove to be an issue. However, many of the limitations deterring demand can in fact be opportunities for marine PES schemes. Issues such as tenure and enforcement can be opportunities for local empowerment. However, with this comes a warning of caution. In promoting these benefits, one must not loose sight of the pre-existing exclusions and marginalisation which already occur within artisanal communities. In devolving rights, recognition of local conditions will be needed in order to prevent monopolisation by the more elite. Community contracts should take care not to further marginalise the more vulnerable.

8.5 Conclusions

PES continue to receive a great deal of attention within development and conservation fields alike, and have much to live up to. In the marine environment PES are even more nascent; at present most are in the proposal stage. As previously discussed, there has been little other critical analysis of PES within a coastal and marine context. More worrying is the lack of research associated with possible negative effects within an environment known to house some of the world's most vulnerable.

In light of this, we have identified a number of issues that marine PES will need to address if they are to be successful.

In the first instance, work is needed to secure demand. However, many of the issues relating to the nature of marine environmental services and tenure can be overcome, and indeed should perhaps be looked at as opportunities within the marine and coastal environment to secure rights and improve understanding.

However, in devolving tenure and enforcement rights to communities, marine PES must deal with small-scale communities which have underlying inequities. On top of this, the nature of PES schemes is one that limits the participation of some. Without careful consideration of the most vulnerable, marine PES can exacerbate inequity by preventing participation of some vulnerable actors. Care should be taken in the design of marine PES schemes. In addition to design factors such as eligibility criteria and programme restrictions, the influence that the various types of incentive offered and the inherent community structures should be acknowledged.

One anticipates that PES will continue to flourish within the marine management portfolio, in part because it directly addresses some of the key shortcomings of many of its predecessors. While we present a rosy picture as to how marine PES can overcome some of the more obvious challenges, the final message should be one of caution. Like so many failed instruments before it, the long-term successful of marine PES will be based on a perception of legitimacy. In turn legitimacy will depend on both the effectiveness of management outcomes and the fair and just sharing of benefits. As such, getting it 'socially right' should be of interest not only to those with development agendas but to environmental conservationists as well.

References

Aas, Ø., Haider, W., Hunt, L., 2000. Angler responses to potential harvest regulations in a Norwegian sport fishery: a conjoint-based choice modeling approach. North American Journal of Fisheries Management 20, 940–950.

Adger, W.N., 2003. Social Capital, Collective Action, and Adaptation to Climate Change. Economic Geography 79, 387–404.

Adhikari, B., Boag, G., 2013. Designing payments for ecosystem services schemes: some considerations. Current Opinion in Environmental Sustainability 5, 72-77.

Agardy, T., 2000. Effects of fisheries on marine ecosystems: a conservationist's perspective. ICES Journal of Marine Science 57, 761–765.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J., Dayton, P.K., Kenchington, R., Laffoley, D., McConney, P., Murray, P.A., Parks, J.E., Peau, L., 2003. Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. Aquatic Conserv: Mar. Freshw. Ecosyst. 13, 353–367.

Agrawal, A., 2002. Common resources and institutional sustainability. The Drama of the Commons. National Academy Press, Washington DC, pp. 41–86.

Agarwal, B., 1997. Re-sounding the alert-gender, resources and community action. World Development 25, 1373–1380.

Agarwal, B., 2000. Conceptualising environmental collective action: why gender matters. Cambridge Journal of Economics 24, 283–310.

Agarwal, B., 2001. Participatory Exclusions, Community Forestry, and Gender: An Analysis for South Asia and a Conceptual Framework. World Development 29, 1623–1648.

Agarwal, B., 2009. Gender and forest conservation: The impact of women's participation in community forest governance. Ecological Economics 68, 2785–2799.

Agrawal, A., Yadama, G., Andrade, R., Bhattacharya, A., 2006. Decentralization and environmental conservation: Gender effects from participation in joint forest management. Citeseer.

Aguilar, L., Castaneda, I., 2001. About fishermen, fisherwomen, oceans and tides: a gender perspective in marine-coastal zones. IUCN-ORMA, San Jose, Costa Rica.

Ai, C., Norton, E.C., 2003. Interaction terms in logit and probit models. Economics Letters 80, 123–129.

Akoten, J.E., Sawada, Y., Otsuka, K., 2006. The determinants of credit access and its impacts on micro and small enterprises: The case of garment producers in Kenya. Economic Development and Cultural Change 54, 927–944.

Akpalu, W., 2010. A Dynamic Model of Mesh Size Regulatory Compliance. Journal of Agricultural and Resource Economics 35, 34.

Akpalu, W., 2011a. Fisher skills and compliance with effort-limiting fishing regulations in a developing country: The case of Ghana. International Journal of Social Economics 38, 666–675.

Akpalu, W., 2011b. Determinants of noncompliance with light attraction regulation among inshore fishers in Ghana. The Journal of Socio-Economics 40, 172–177.

Alderman, H., Paxson, C.H., 1992. Do the poor insure? a synthesis of the literature on risk and consumption in developing countries, in: Economics in a Changing World: Development, Trade and the Environment, Proceedings of the Tenth World Congress of the International Economic Association. St Martin's Press, Moscow, pp. 49–78.

Aldon, M.E.T., Fermin, A.C., Agbayani, R.F., 2011. Socio-cultural context of fishers' participation in coastal resources management in Anini-y, Antique in west central Philippines. Fisheries Research 107, 112–121.

Alix-Garcia, J., de Janvry, A., Sadoulet, E., 2005. The role of risk in targeting payments for environmental services. University of Montanta, Missoula. Available at SSRN: http://ssrn.com/abstract=836144 or http://dx.doi.org/10.2139/ssrn.836144

Alix-Garcia, J.M., Shapiro, E.N., Sims, K.R., 2012. Forest conservation and slippage: Evidence from Mexico's national payments for ecosystem services program. Land Economics 88, 613–638.

Allison, E.H., Horemans, B., 2006. Putting the principles of the Sustainable Livelihoods Approach into fisheries development policy and practice. Marine Policy 30, 757–766.

Allison, E.H., 2001. The livelihoods approach and management of small-scale fisheries. Marine Policy 25, 377–388.

Alpízar, F., Nordén, A., Pfaff, A., Robalino, J., 2013. Effects of Exclusion from a Conservation Policy: Negative Behavioral Spillovers from Targeted Incentives. Environment. Duke Environmental and Energy Economics Working Paper Series No. EE 13-06, 28pp.

Alston, L.J., Andersson, K., Smith, S.M., 2013. Payment for Environmental Services: Hypotheses and Evidence. Annual Review of Resource Economics 5, 139–159.

Angelsen, A., 2008. Moving ahead with REDD: issues, options and implications. CIFOR. Available at: http://www.cifor.org/online-library/browse/view-publication/publication/2601.html . Accessed 23/11/2013.

Atkinson, G., Mourato, S., 2008. Environmental cost-benefit analysis. Annual Review of Environment and Resources 33, 317–344.

Baden, S., Reeves, H., 2000. Gender and development: Concepts and definitions. Institute of Development Studies University of Sussex.

Bagachwa, M.S.D., Hodd, M.R.V., Maliyamkono, T.L., others, 1994. Fisheries and development in Tanzania. Macmillan Press Ltd.

Balmford, A., Gaston, K.J., Blyth, S., James, A., Kapo, V., 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. PNAS 100, 1046–1050.

Balmford, A., Gravestock, P., Hockley, N., McClean, C.J., Roberts, C.M., 2004. The worldwide costs of marine protected areas. PNAS 101, 9694–9697.

Balmford, A., Whitten, T., 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx 37, 238–250.

Bandiera, O., Rasul, I., 2003. Complementarities, social networks, and technology adoption in Mozambique. Retrieved September 6, 2004.

Bandiera, O., Rasul, I., 2006. Social Networks and Technology Adoption in Northern Mozambique. Economic Journal 116, 869–902.

Barbier, E.B., 2010. Progress and challenges in valuing coastal and marine ecosystem services. Review of Environmental Economics and Policy 6, 1–19.

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.V., Bael, D., Kappel, C.V., Perillo, G.M.E. & Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. science 319, 321–323.

Barbier, E.B., Tesfaw, A.T., 2012. Can REDD+ save the forest? The role of payments and tenure. Forests 3, 881–895.

Barr, R.F., 2010. Conditional Cash Transfers in the coastal context of Mtwara Development Corridor, Tanzania: economic and livelihood analysis. CARE International in association with WWF.

Barr, R.F., Mourato, S., 2009. Investigating the potential for marine resource protection through environmental service markets: An exploratory study from La Paz, Mexico. Ocean & Coastal Management 52, 568–577.

Barrett, C.B., Reardon, T., Webb, P., 2001. Nonfarm income diversification and household livelihood strategies in rural Africa: concepts, dynamics, and policy implications. Food Policy, 26(4), 315-331

Bateman, I.J., Carson, R.T., Day, B., Hanemann, M., Hanleys, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Ozdemiroglu, E., Pearce, D., Sugden, R., Swanson, J., 2002. Economic valuation with stated preference techniques: a manual. Edward Elgar.

Baticados, D.B., 2004. Fishing cooperatives' participation in managing nearshore resources: the case in Capiz, central Philippines. Fisheries research 67, 81–91.

Baumgart-Getz, A., Prokopy, L.S., Floress, K., 2012. Why farmers adopt best management practice in the United States: a meta-analysis of the adoption literature. Journal of environmental management 96, 17–25.

Beger, M., Harborne, A.R., Dacles, T.P., Solandt, J.-L., Ledesma, G.L., 2005. A framework of lessons learned from community-based marine reserves and its effectiveness in guiding a new coastal management initiative in the Philippines. Environmental Management 34, 786–801.

Begossi, A., May, P.H., Lopes, P.F., Oliveira, L.E.C., da Vinha, V., Silvano, R.A.M., 2011. Compensation for environmental services from artisanal fisheries in SE Brazil: Policy and technical strategies. Ecological Economics 71, 25–32.

Béné, C., 2009. Are Fishers Poor or Vulnerable? Assessing Economic Vulnerability in Small-Scale Fishing Communities. Journal of Development Studies 45, 911.

Béné, C., 2003. When fishery rhymes with poverty: a first step beyond the old paradigm on poverty in small-scale fisheries. World Development 31, 949–975.

Béné, C., Mindjimba, K., Belal, E., Jolley, T., 2000. Evaluating livelihood strategies and the role of inland fisheries in rural development and poverty alleviation: the case of the Yaéré Floodplain in North Cameroon. IIFET 2000 Proceedings.

Béné, C., Hersoug, B., Allison, E.H., 2010. Not by Rent Alone: Analysing the Pro-Poor Functions of Small-Scale Fisheries in Developing Countries. Development Policy Review 28, 325–358.

Benítez, P.C., Kuosmanen, T., Olschewski, R., Van Kooten, G.C., 2006. Conservation payments under risk: a stochastic dominance approach. American Journal of Agricultural Economics 88, 1–15.

Bennett, E., 2005. Gender, fisheries and development. Marine Policy 29, 451–459.

Bennett, E., Valette, H.R., Mäiga, K.Y., Medard, M., 2004. Room to manoevre: gender and coping strategies in the fsheries sector: The report of a workshop help in Cotonou, Benin 1-4 December 2003. IDDRA, Portmouth.

Bennett, G., Carroll, N., Hamilton, K., 2013. Charting new waters: State of watershed payments 2012. Ecosystem Marketplace, Washington, DC.

Bennett, J., Blamey, R., 2001. The choice modelling approach to environmental valuation. Edward Elgar Publishing.

Bennett, M.T., 2008. China's sloping land conversion program: institutional innovation or business as usual? Ecological Economics 65, 609–711.

Bergseng, E., Vatn, A., 2009. Why protection of biodiversity creates conflict–Some evidence from the Nordic countries. Journal of Forest Economics 15, 147–165.

Berkes, F., 2003. Alternatives to conventional management: Lessons from small-scale fisheries. Environments 31, 5–20.

Berkes, F., Mahon, R., McConney, P., Pollnac, R., Pomeroy, R., 2001. Managing Small-scale Fisheries: Alternative directions and methods. The International Development Research Centre (IDRC), Ottawa.

Bess, R., Rallapudi, R., 2007. Spatial conflicts in New Zealand fisheries: The rights of fishers and protection of the marine environment. Marine Policy 31, 719–729.

Bjørnskov, C., 2006. The multiple facets of social capital. European Journal of Political Economy 22, 22–40.

Blom, B., Sunderland, T., Murdiyarso, D., 2010. Getting REDD to work locally: lessons learned from integrated conservation and development projects. Environmental Science & Policy, 13, 164-172.

Bodin, Ö., Crona, B.I., 2008. Management of natural resources at the community level: exploring the role of social capital and leadership in a rural fishing community. World Development 36, 2763–2779.

Borja, A., Bricker, S.B., Dauer, D.M., Demetriades, N.T., Ferreira, J.G., Forbes, A.T., Hutchings, P., Jia, X., Kenchington, R., Marques, J.C., Zhu, C., 2008. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Marine Pollution Bulletin 56, 1519–1537.

Boxall, P., Adamowicz, W.L. (Vic), Moon, A., 2009. Complexity in choice experiments: choice of the status quo alternative and implications for welfare measurement*. Australian Journal of Agricultural and Resource Economics 53, 503–519.

Breetz, H.L., Fisher-Vanden, K., Jacobs, H., Schary, C., 2005. Trust and communication: Mechanisms for increasing farmers' participation in water quality trading. Land Economics 81, 170–190.

Bulte, E.H., Lipper, L., Stringer, R., Zilberman, D., 2008. Payments for ecosystem services and poverty reduction: concepts, issues, and empirical perspectives. Environment and Development Economics 13, 245–254.

Bunce, M., Rosendo, S., Brown, K., 2010. Perceptions of climate change, multiple stressors and livelihoods on marginal African coasts. Environment, Development and Sustainability 12, 407–440.

Cacho, O.J., Marshall, G.R., Milne, M., 2005. Transaction and abatement costs of carbon-sink projects in developing countries [WWW Document].

http://econpapers.repec.org/article/cupendeec/v_3a10_3ay_3a2005_3ai_3a05_3ap_3a597-614_5f00.htm (accessed 8.19.09).

Carlsson, L., Berkes, F., 2005. Co-management: concepts and methodological implications. Journal of environmental management 75, 65–76.

Carney, J., 1993. Converting the wetlands, engendering the environment: The intersection of gender with agrarian change in The Gambia. Reading Economic Geography 220–234.

Carr, M.H., Neigel, J.E., Estes, J.A., Andelman, S., Warner, R.R., Largier, J.L., 2003. Comparing marine and terrestrial ecosystems: implications for the design of coastal marine reserves. Ecological Applications 13, 90–107.

Charmaz, K., 2006. Constructing grounded theory: A practical guide through qualitative analysis. Sage Publications Ltd.

Chen, X., Lupi, F., He, G., Liu, J., 2009. Linking social norms to efficient conservation investment in payments for ecosystem services. Proc Natl Acad Sci U S A 106, 11812–11817.

Chhatre, A., Agrawal, A., 2008. Forest commons and local enforcement. Proceedings of the National Academy of Sciences 105, 13286–13291.

Cho, L., 2005. Marine protected areas: a tool for integrated coastal management in Belize. Ocean & Coastal Management 48, 932–947.

Christie, P., 2004. Marine protected areas as biological successes and social failures in Southeast Asia. American Fisheries Society Symposium 42, 155–164.

Christie, P., White, A.T., 2007. Best practices for improved governance of coral reef marine protected areas. Coral Reefs 26, 1047–1056.

Christy, F.T., 1982. Territorial use rights in marine fisheries: definitions and conditions. Food & Agriculture Organisation.

Cinner, J.E., 2010. Poverty and the Use of Destructive Fishing Gear Near East African Marine Protected Areas. Environmental Conservation 36, 321–326.

Cinner, J.E., Daw, T., McClanahan, T.R., 2009. Socioeconomic factors that affect artisanal fishers' readiness to exit a declining fishery. Conservation Biology 23, 124–130.

Cinner, J.E., McClanahan, T.R., Graham, N.A.J., Pratchett, M.S., Wilson, S.K., Raina, J., 2009. Gear-based fisheries management as a potential adaptive response to climate change and coral mortality. Journal of Applied Ecology 46, 724–732.

Cinner, J.E., Pollnac, R., 2004. Poverty, perceptions and planning: why socioeconomics matter in the management of Mexican reefs. Ocean & Coastal Management 47, 479–493.

Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., García-Charton, J.-A., Pérez-Ruzafa, Á., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F., Culioli, J.-M., Dimech, M., Falcón, J.M., Guala, I., Milazzo, M., Sánchez-Meca, J., Somerfield, P.J., Stobart, B., Vandeperre, F., Valle, C., Planes, S., 2008. Marine reserves: size and age do matter. Ecology Letters 11, 481–489.

Clay, D., Reardon, T., Kangasniemi, J., 1998. Sustainable intensification in the highland tropics: Rwandan farmers' investments in land conservation and soil fertility. Economic Development and Cultural Change 46, 351–377.

Clements, T., John, A., Nielsen, K., An, D., Tan, S., Milner-Gulland, E.J., 2010. Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. Ecological Economics 69, 1283–1291.

Coady, D., Grosh, M., Hoddinott, J., 2004. Targeting outcomes redux. World Bank Research Observer 19, 61–85.

Coase, R., 1960. The problem of social cost. The Journal of Law and Economics 3, 1–44.

Coate, S., Ravallion, M., 1993. Reciprocity without commitment : Characterization and performance of informal insurance arrangements. Journal of Development Economics 40, 1–24.

Conley, T.G., Udry, C.R., 2010. Learning about a new technology: Pineapple in Ghana. The American Economic Review 100, 35–69.

Cook, K.S., Hardin, R., Levi, M., 2007. Cooperation without trust? Russell Sage Foundation.

Cooper, J.C., 2003. A joint framework for analysis of agri-environmental payment programs. American journal of agricultural economics 85, 976–987.

Corbera, E., Brown, K., 2010. Offsetting benefits? Analyzing access to forest carbon. Environment and planning. A 42, 1739.

Corbera, E., Brown, K., Adger, W.N., 2007a. The equity and legitimacy of markets for ecosystem services. Development and Change 38, 587–613.

Corbera, E., Kosoy, N., Tuna, M.M., 2007b. Equity implications of marketing ecosystem services in protected areas and rural communities: case studies from Meso-America. Global Environmental Change 17, 365–380.

Corbera, E., Pascual, U., 2012. Ecosystem services: heed social goals. Science 335, 655–656.

Corbin, J., Strauss, A., 2008. Basics of qualitative research: Techniques and procedures for developing grounded theory, 3rd ed. Sage Publications, Inc., Thousand Oaks, CA.

Cornwall, A., 2003. Whose Voices? Whose Choices? Reflections on Gender and Participatory Development. World Development 31, 1325–1342.

Costello, C., Gaines, S.D., Lynham, J., 2008. Can catch chares prevent fisheries collapse. Science 321.

Coulthard, S., 2012. What does the debate around social wellbeing have to offer sustainable fisheries? Current Opinion in Environmental Sustainability 4, 358–363.

Crawford, B.R., Siahainenia, A., Rotinsulu, C., Sukmara, A., 2004. Compliance and enforcement of community-based coastal resource management regulations in North Sulawesi, Indonesia. Coastal Management, 32, 39–50.

Crona, B., Bodin, Ö., 2006. What you know is who you know? Communication patterns among resource users as a prerequiste for co-management. Ecology and Society 11, 7.

Curnock, M., 2010. Mechanisms for assessing the sustainability of swimming-with-whales tourism in the Great Barrier Reef. PhD thesis, James Cook University.

D'Emden, F.H., Llewellyn, R.S., Burton, M.P., 2008. Factors influencing adoption of conservation tillage in Australian cropping regions*. Australian Journal of Agricultural and Resource Economics 52, 169–182.

D'Souza, G., Cyphers, D., Phipps, T., 1993. Factors affecting the adoption of sustainable agricultural practices. Agricultural and Resource Economics Review 22, 159–165.

Dadi, O. 2010. Mtwara District Fisheries Office. *Pers comm.*

Dahdouh-Guebas, F., Jayatissa, L.P., Di Nitto, D., Bosire, J.O., Lo Seen, D., Koedam, N., 2005. How effective were mangroves as a defence against the recent tsunami? Current Biology 15, R443–R447.

Danielsen, F., Skutsch, M., Burgess, N.D., Jensen, P.M., Andrianandrasana, H., Karky, B., Lewis, R., Lovett, J.C., Massao, J., Ngaga, Y., Phartiyal, P., Poulsen, M.K., Singh, S.P., Solis, S., Sørensen, M., Tewari, A., Young, R., Zahabu, E., 2011. At the heart of REDD+: a role for local people in monitoring forests? Conservation Letters 4, 158–167.

Dasgupta, P., Serageldin, I., 2001. Social capital: a multifaceted perspective. World Bank Publications.

Datta, S.K., Sarkar, K., 2010. Status of joint forest management in india: Socio-economic determinants of forest participation in a dynamic optimization setting. Int. J. Soc. For 3, 81–100.

Davis, D., Gartside, D.F., 2001. Challenges for economic policy in sustainable management of marine natural resources. Ecological Economics 36, 223–236.

Daw, T.M., 2008. Spatial distribution of effort by artisanal fishers: Exploring economic factors affecting the lobster fisheries of the Corn Islands, Nicaragua. Fisheries Research 90, 17–25.

Daw, T.M., Cinner, J.E., McClanahan, T.R., Brown, K., Stead, S.M., Graham, N.A.J., Maina, J., 2012. To Fish or Not to Fish: Factors at Multiple Scales Affecting Artisanal Fishers' Readiness to Exit a Declining Fishery. PLoS ONE 7, e31460.

Day, J.C., 2002. Zoning—lessons from the Great Barrier Reef Marine Park. Ocean & Coastal Management 45, 139–156.

De Janvry, A., Fafchamps, M., Sadoulet, E., 1991. Peasant household behaviour with missing markets: some paradoxes explained. The Economic Journal 101, 1400–1417.

De Janvry, A., Sadoulet, E., 2006. Progress in the modeling of rural households' behavior under market failures, in: Poverty, Inequality and Development. Springer, pp. 155–181.

De Janvry, A., Sadoulet, E., Solomon, P., Vakis, R., 2006. Uninsured risk and asset protection: Can conditional cash transfer programs serve as safety nets. The World Bank, Social Protection Working Paper.

De Silva, D., 2011. Faces of women in global fishery value chains: Female involvement, impact and importance in fisheries of developed and developing countries. Report produced for NORAD/FAO Chain Project, Food and Agriculture Organization of the United Nations, pp.49.

De Walque, D., Dow, W.H., Nathan, R., Abdul, R., Abilahi, F., Gong, E., Isdahl, Z., Jamison, J., Jullu, B., Krishnan, S., 2012. Incentivising safe sex: a randomised trial of conditional cash transfers for HIV and sexually transmitted infection prevention in rural Tanzania. BMJ open 2.

Defeo, O., Castilla, J.C., 2005. More than one bag for the world fishery crisis and keys for co-management successes in selected artisanal Latin American shellfisheries. Reviews in Fish Biology and Fisheries 15, 265–283.

Defrancesco, E., Gatto, P., Runge, F., Trestini, S., 2008. Factors Affecting Farmers' Participation in Agri-environmental Measures: A Northern Italian Perspective. Journal of Agricultural Economics 59, 114–131.

Delacote, P., Garcia, S., Stenger, A., Tu, G., undated. Private forest owners' participation behavior related to an incentive conservation program: A case study of Natura 2000 contracts in France. Working Paper INRA, UMR 356 Laboratoire d'Economie Forestiere.

Delacote, P., Palmer, C., Bakkegaard, R.K., Thorsen, B.J., 2013. Unveiling information on opportunity costs in REDD: Who obtains the surplus when policy objectives differ? Resource and Energy Economics. In press.

Depondt, F., Green, E., 2006. Diving user fees and the financial sustainability of marine protected areas: opportunities and impediments. Ocean & Coastal Management 49, 188–202.

Dercon, S., 2002. Income Risk, Coping Strategies, and Safety Nets. The World Bank Research Observer 17, 141 –166.

Dercon, S., Christiaensen, L., 2011. Consumption risk, technology adoption and poverty traps: evidence from Ethiopia. Journal of Development Economics 96, 159–173.

Deressa, T.T., Hassan, R.M., Ringler, C., Alemu, T., Yesuf, M., 2009. Determinants of farmers' choice of adaptation methods to climate change in the Nile Basin of Ethiopia. Global Environmental Change 19, 248–255.

Derissen, S., Quaas, M.F., 2013. Combining performance-based and action-based payments to provide environmental goods under uncertainty. Ecological Economics 85, 77–84.

Di Falco, S., Bulte, E., 2011. A dark side of social capital? Kinship, consumption and savings. Journal of Development Studies 47, 1128–1151.

Di Falco, S., Bulte, E., Yesuf, M., 2010. Social capital and weather shocks in Ethiopia: Climate change and culturally-induced poverty traps, in: Centre for the Study of African Economies Conference, Edinburgh, UK.

Diagne, A., 1999. Determinants of household access to and participation in formal and informal credit markets in Malawi. Food Consumption and Nutrition Division discussion paper 67. International Food Policy Research Institute. Washington DC.

Dixon, C.J., Leach, B., 1982. Sampling methods for geographical research. CAT MOG 17.

Dobbs, T.L., Pretty, J., 2008. Case study of agri-environmental payments: the United Kingdom. Ecological Economics 65, 765–775.

Dolisca, F., Carter, D.R., McDaniel, J.M., Shannon, D.A., Jolly, C.M., 2006. Factors influencing farmers' participation in forestry management programs: A case study from Haiti. Forest Ecology and Management 236, 324–331.

Dorussen, H., Lenz, H., Blavoukos, S., 2005. Assessing the reliability and validity of expert interviews. European Union Politics 6, 315–337.

Dunlap, R.E., Van Liere, K.D., 1984. Commitment to the dormant social paradigm and concern for environmental quality. Social Science Quarterly 65, 1012–1028.

Dutschke, M., Angelsen, A., 2008. How do we ensure permanence and assign liability? Center for International Forestry Research (CIFOR). Available at: http://www.cifor.org/nc/online-library/browse/view-publication/publication/2668.html (Accessed 14/11/2013)

Echavarría, M., Vogel, J., Albán, M., Meneses, F., 2003. The Impacts of Payments for Watershed Services in Ecuador: Emerging lessons from Pimampiro and Cuenca. International Institute for Environmental and Development, London.

Edinger, E.N., Jompa, J., Limmon, G.V., Widjatmoko, W., Risk, M.J., 1998. Reef degradation and coral biodiversity in Indonesia: Effects of land-based pollution, destructive fishing practices and changes over time. Marine Pollution Bulletin 36, 617–630.

Edmund, J., 2008. Incorporating Gender into PHE Strategies: Experiences from Conservation International. Conservation International, Washington D.C.

Ellis, F., 2000. The Determinants of Rural Livelihood Diversification in Developing Countries. Journal of Agricultural Economics 51, 289–302.

Emtage, N., Herbohn, J., 2012. Implications of landholders' management goals, use of information and trust of others for the adoption of recommended practices in the Wet Tropics region of Australia. Landscape and Urban Planning 107, 351-360.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: an overview of the issues. Ecological Economics 65, 663–674.

Engel, S., Palmer, C., 2008. Payments for environmental services as an alternative to logging under weak property rights: the case of Indonesia. Ecological Economics 65, 799–809.

Espinosa-Goded, M., Barreiro-Hurlé, J., Ruto, E., 2010. What Do Farmers Want From Agri-Environmental Scheme Design? A Choice Experiment Approach. Journal of Agricultural Economics 61, 259–273.

Evans, D., 2008. Tanzania Community-Based Conditional Cash Transfer (CB-CCT) Pilot. Available from: http://gssp.newamerica.net/sites/gssp.newamerica.net/files/articles/Tanzania%20Community%20Based%20CCT%20Pilot.pdf (Accessed 01/01/2014)

Evans, D., Kosec, K., De Weerdt, J., 2012. Tanzania Community Based Conditional Cash Transfer Program, in: Workshop on Evaluating the Impact of Cash Transfer Programs in Sub Saharan Africa. Presented at the The Transfer Project, The World Bank.

Evans, T., Arpels, M., Clements, T., 2012. Pilot REDD Activities in Cambodia are Expected to Improve Access to Forest Resource Use Rights and Land Tenure for Local Communities., in: Naughton-Treves, L. and Day, C. (eds) Lessons about Land Tenure, Forest Governance and REDD+. Case Studies from Africa, Asia and Latin America. UW-Madison Land Tenure Center, Madison, Wisconsin, pp. 73–82.

Evans, T.G., Adams, A.M., Mohammed, R., Norris, A.H., 1999. Demystifying Nonparticipation in Microcredit: A Population-Based Analysis. World Development 27, 419–430.

Ezemenari, K., Chaudhury, N., Owens, J., 2002. Gender and risk in the design of social protection interventions. World Bank.

Fafchamps, M., Lund, S., 2003. Risk-sharing networks in rural Philippines. Journal of Development Economics 71, 261–287.

Falconer, K., 2000. Farm-level constraints on agri-environmental scheme participation: a transactional perspective. Journal of rural studies 16, 379–394.

FAO, 2001. Cultural characteristics of small-scale fishing communities, in: Understanding the Cultures of Fishing Communities: A Key to Fisheries Management and Food Security, FAO Fisheries Technical Paper. p. 287.

FAO, 2005a. Increasing the contribution of small-scale fisheries to poverty alleviation and food security. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2005b. Fishers number double since 1970. Available at: ftp://ftp.fao.org/fi/DOCUMENT/c929_article/C929e_article.pdf [Accessed 10/1/2009].

FAO, 2007. The state of food and agriculture: paying farmers for environmental services. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2008. The Sunken Billions: The Economic Justification for Fisheries Reform. The International Bank for Reconstruction and Development / World Bank, Washington, DC.

FAO, 2009. The State of World Fisheries and Aquaculture 2008. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2010. The State of World Fisheries and Aquaculture. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2012. The State of World Fisheries and Aquaculture - 2012. FAO Fisheries and Aquaculture Department, Food and Agriculture Organisation of the United Nations, Rome.

FAO, n.d. National fishery sector overview: The United Republic of Tanzania [WWW Document]. Available at: http://www.fao.org/fishery/countrysector/naso_tanzania/en [Accessed 10/1/2009].

FAO, World Fish Center, 2008. Small-scale capture fisheries: A gloable overview with emphasis on developing countries. A premilinary report of the Big Numbers Project. Food and Agriculture Organisation of the United Nations, Rome.

Fazey, I., Fazey, J.A., Salisbury, J.G., Lindenmayer, D.B., Dovers, S., 2006. The nature and role of experiential knowledge for environmental conservation. Environmental Conservation 33, 1–10.

Ferraro, P.J., 2008. Asymmetrical information and contract design for payments for environmental services. Ecological Economics 65, 810–821.

Ferraro, P.J., 2001. Global habitat protection: limitations of development interventions and a role for conservation performance payments. Conservation Biology 15, 990–1000.

Ferraro, P.J., Kiss, A., 2002. Direct payments to conserve biodiversity. Science 298, 1718–1719.

Ferraro, P.J., Simpson, R.D., 2002. The cost-effectiveness of conservation payments. Land Economics 78, 339–353.

Ferse, S.C.A., Costa, M.M., Máñez, K.S., Adhuri, D.S., Glaser, M., 2010. Allies, not aliens: increasing the role of local communities in marine protected area implementation. Environmental Conservation 37, 23–34.

Fischer, E., Qaim, M., 2011. Smallholder Farmers and Collective Action: What Determines the Intensity of Participation? in: European Association of Agricultural Economists International Congress, August 2011, Zurich, Switzerland.

Fisher, B., 2012. Poverty, Payments, and Ecosystem Services in the Eastern Arc Mountains of Tanzania, in: Ingram, J.C., DeClerck, F., Rio, C.R. del (Eds.), Integrating Ecology and Poverty Reduction. Springer New York, pp. 173–189.

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R.K., Burgess, N.D., 2010. Common pool resource management and PES: Lessons and constraints for water PES in Tanzania. Ecological Economics 69, 1253–1261.

Fisher, B., Swetnam, R., Burgess, N., McKenzie, E., Wilcock, S., 2012. InVEST scenarios case study: Eastern Arc Mountains, Tanzania. WWF - Natural Capital Project.

Forest Trends & The Katoomba Group, 2010. Payments for Ecosystem Services: Getting started in marine and coastal ecosystems. A Primer. Forest Trends & The Katoomba Group.

Francis, J., Bryceson, I., 2000. Tanzanian coastal and marine resources: some examples illustrating questions of sustainable use. In: Lessons learned: Case studies in sustainable use. IUCN, 76-102

Frewer, L., 1999. Risk perception, social trust, and public participation in strategic decision making: Implications for emerging technologies. Ambio 569–574.

Frost, P.G.H., Bond, I., 2008. The CAMPFIRE programme in Zimbabwe: Payments for wildlife services. Ecological Economics 65, 776–787.

Gabunda, F.E.M., Barker, R., 1994. Adoption of contour hedgerows in Matalom, Leyte, Philippines, in: Workshop on Innovations and Social Change: Who Is Empowered.

Game, E.T., Grantham, H.S., Hobday, A.J., Pressey, R.L., Lombard, A.T., Beckley, L.E., Gjerde, K., Bustamante, R., Possingham, H.P., Richardson, A.J., 2009. Pelagic protected areas: the missing dimension in ocean conservation. Trends in ecology & evolution 24, 360–369.

Gauvin, C., Uchida, E., Rozelle, S., Xu, J., Zhan, J., 2010. Cost-effectiveness of payments for ecosystem services with dual goals of environment and poverty alleviation. Environmental Management 45, 488–501.

Gebremedhin, B., Swinton, S.M., 2003. Investment in soil conservation in northern Ethiopia: the role of land tenure security and public programs. Agricultural Economics 29, 69–84.

Geheb, K., Crean, K., 2003. Community-level access and control in the management of Lake Victoria's fisheries. Journal of Environmental Management 67, 99–106.

Gelcich, S., Edwards-Jones, G., Kaiser, M.J., 2005. Importance of Attitudinal Differences among Artisanal Fishers toward Co-Management and Conservation of Marine Resources. Conservation Biology 19, 865–875.

Gelcich, S., Godoy, N., Castilla, J.C., 2009. Artisanal fishers' perceptions regarding coastal co-management policies in Chile and their potentials to scale-up marine biodiversity conservation. Ocean & Coastal Management 52, 424–432.

Gelcich, S., Godoy, N., Prado, L., Castilla, J.C., 2008. Add-on conservation benefits of marine territorial user rights fishery policies in central Chile. Ecological Applications 18, 273–281.

Gell, F.R., Roberts, C.M., 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends in Ecology & Evolution 18, 448–455.

Gibbons, J.M., Nicholson, E., Milner-Gulland, E.., Jones, J.P.., 2011. Should payments for biodiversity conservation be based on action or results? Journal of Applied Ecology 48, 1218-1226.

Gibson, C.C., Williams, J.T., Ostrom, E., 2005. Local enforcement and better forests. World Development 33, 273–284.

Giri, K., Darnhofer, I., 2010. Nepali women using community forestry as a platform for social change. Society and Natural Resources 23, 1216–1229.

Gladwin, C.H., Peterson, J.S., Mwale, A.C., 2002. The quality of science in participatory research: A case study from Eastern Zambia. World Development 30, 523–543.

Glenk, K., Barkmann, J., Marggraf, R., 2006. Locally Perceived Values of Biological Diversity in Indonesia–a Choice Experiment Approach, in: 8th Annual BIOECON Conference on Economic Analysis of Ecology and Biodiversity. pp. 15–29.

Godquin, M., Quisumbing, A.R., 2008. Separate but equal? The gendered nature of social capital in rural Philippine communities. Journal of International Development 20, 13–33.

Goldman-Benner, R.L., Benitez, S., Boucher, T., Calvache, A., Daily, G., Kareiva, P., Kroeger, T., Ramos, A., 2012. Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. Oryx 46, 55–63.

Gong, Y., Bull, G., Baylis, K., 2010. Participation in the world's first clean development mechanism forest project: the role of property rights, social capital and contractual rules. Ecological Economics 69, 1292–1302.

González, J.R., Kolehmainen, O., Pukkala, T., 2007. Using expert knowledge to model forest stand vulnerability to fire. Computers and Electronics in Agriculture 55, 107–114.

Grafton, R.Q., 2005. Social capital and fisheries governance. Ocean & Coastal Management 48, 753–766.

Graham, N.A.J., McClanahan, T.R., MacNeill, M.A., Wilson, S.K., Polunin, N.V.C., Jennings, S., Chabanet, P., Clark, S., Spalding, M.D., Letourneur, Y., Bigot, L., Galzin, R., Ohman, M.C., Garpe, K.C., Edwards, A.J., Sheppard, C.R.C., 2008. Climate Warming, Marine Protected Areas and the Ocean-Scale Integrity of Coral Reef Ecosystems. PLoS ONE 3, e3039. doi:10.1371/journal.pone.0003039.

Granger Morgan, M., Pitelka, L., Shevliakova, E., 2001. Elicitation of Expert Judgments of Climate Change Impacts on Forest Ecosystems. Climatic Change 49, 279–307.

Granovetter, M., 1985. Economic action and social structure: the problem of embeddedness. American Journal of Sociology 91, 481–510.

Granovetter, M., 2005. The impact of social structure on economic outcomes. Journal of Economic Perspectives 19, 33–50.

Grieg-Gran, M., Porras, I., Wunder, S., 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Development 33, 1511–1527.

Grimsditch, G., Kenchington, R., Alder, J., Tamelander, J., Nakamura, T., 2012. The blue carbon special edition–introduction and overview. Ocean & Coastal Management. (in press) Available at: http://www.sciencedirect.com/science/article/pii/S0964569112000981 [Accessed September 5, 2012].

Groom, B., 2012. The Impact of the SLCP on Income and Poverty Alleviation. Environmental Economy and Policy Research Discussion Paper Series, N. #.2011 Land Economy Department. University of Cambridge.

Groom, B., Grosjean, P., Kontoleon, A., Swanson, T., Zhang, S., 2010. Relaxing rural constraints: a "win-win"policy for poverty and environment in China? Oxford Economic Papers 62, 132–156.

Groom, B., Palmer, C., 2010. Cost-effective provision of environmental services: the role of relaxing market constraints. Environment and Development Economics 15, 219.

Groom, B., Palmer, C., 2012a. REDD+ and rural livelihoods. Biological Conservation 154, 42–52.

Groom, B., Palmer, C., 2012b. Relaxing constraints as a conservation policy. FEEM Working Paper No. 63.2012. Available at SSRN: http://ssrn.com/abstract=2161807 or http://dx.doi.org/10.2139/ssrn.2161807

Gross-Camp, N.D., Martin, A., McGuire, S., Kebede, B., Munyarukaza, J., 2012. Payments for ecosystem services in an African protected area: exploring issues of legitimacy, fairness, equity and effectiveness. Oryx 46, 24–33.

Guerreiro, J., Chircop, A., Dzidzornu, D., Grilo, C., Ribeiro, R., van der Elst, R., Viras, A., 2011. The role of international environmental instruments in enhancing transboundary marine protected areas: An approach in East Africa. Marine Policy 35, 95–104.

Guerreiro, J., Chircop, A., Grilo, C., Viras, A., Ribeiro, R., van der Elst, R., 2010. Establishing a transboundary network of marine protected areas: Diplomatic and management options for the east African context. Marine Policy 34, 896–910.

Gustavson, K., Kroeker, Z., Walmsley, J., Juma, S., 2009. A process framework for coastal zone management in Tanzania. Ocean & Coastal Management 52, 78–88.

Hagerman, S., Dowlatabadi, H., Satterfield, T., McDaniels, T., 2010. Expert views on biodiversity conservation in an era of climate change. Global Environmental Change 20, 192–207.

Halpern, B.S., 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological applications 13, 117–137.

Halpern, B.S., Longo, C., Hardy, D., McLeod, K.L., Samhouri, J.F., Katona, S.K., Kleisner, K., Lester, S.E., O'Leary, J., Ranelletti, M., Rosenberg, A.A., Scarborough, C., Selig, E.R., Best, B.D., Brumbaugh, D.R., Chapin, F.S., Crowder, L.B., Daly, K.L., Doney, S.C., Elfes, C., Fogarty, M.J., Gaines, S.D., Jacobsen, K.I., Karrer, L.B., Leslie, H.M., Neeley, E., Pauly, D., Polasky, S., Ris, B., St Martin, K., Stone, G.S., Sumaila, U.R., Zeller, D., 2012. An index to assess the health and benefits of the global ocean. Nature 488, 615–620.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., others, 2008. A global map of human impact on marine ecosystems. Science 319, 948–952.

Hamilton, K., Sjardin, M., Peters-Stanley, M., Marcello, T., 2010. Building Bridges: State of the Voluntary Carbon Markets 2010. Ecosystem Marketplace and Bloomberg New Energy Finance, London.

Hanley, N., MacMillan, D., Patterson, I., Wright, R.E., 2003. Economics and the design of nature conservation policy: a case study of wild goose conservation in Scotland using choice experiments. Animal Conservation 6, 123–129.

Hanley, N., Mourato, S., Wright, R.E., 2001. Choice Modelling Approaches: A Superior Alternative for Environmental Valuatioin? Journal of economic surveys 15, 435–462.

Hanley, N., Wright, R.E., Adamowicz, V., 1998. Using choice experiments to value the environment: design issues, current experience and future prospects. Environmental and Resource Economics 11, 413–428.

Hargreaves-Allen, V., Mourato, S., Milner-Gulland, E.J., 2011. A global evaluation of coral reef management performance: are MPAs producing conservation and socio-economic improvements? Environmental Management 47, 684–700.

Harper, S., Zeller, D., Hauzer, M., Pauly, D., Sumaila, U.R., 2013. Women and fisheries: Contribution to food security and local economies. Marine Policy 39, 56–63.

Harrison, E., 2000. Gender, rights and poverty issues: lessons for the sector. Background paper for DFID/FGRP-3/ARP Workshop on: Practical strategies for poverty targeted research. Overseas Development Institute, Hanoi, Vietnam.

Harrison, P., Laizer, J., Grubb, C., 2010. Shifting marine resource incentives: conditional cash tranfers in the coastal context of Mtwara Development Corridor, Mnazi Ba Ruvuma Estuary vicinity, Tanzania. Kilimanyika Ltd.

Hausman, J., McFadden, D., 1984. Specification tests for the multinomial logit model. Econometrica: Journal of the Econometric Society 1219–1240.

Hawkins, J.P., Roberts, C.M., 2004. Effects of artisinal fishing on Caribbean coral reefs. Conservation Biology 18, 215–226.

Hegde, R., Bull, G.Q., 2011. Performance of an agro-forestry based Payments-for-Environmental-Services project in Mozambique: A household level analysis. Ecological Economics 71, 122–130.

Hein, L., Miller, D.C., de Groot, R., 2013. Payments for ecosystem services and the financing of global biodiversity conservation. Current Opinion in Environmental Sustainability 5, 87-93.

Heiss, F., 2002. Structural choice analysis with nested logit models. The Stata Journal 2, 227–252.

Hensher, D.A., Rose, J.M., Greene, W.H., 2005. Applied choice analysis: a primer. Cambridge Univ Pr.

Hockings, M., 2003. Systems for Assessing the Effectiveness of Management in Protected Areas. BioScience 53, 823–832.

Hoffman, A.M., 2002. A Conceptualization of Trust in International Relations. European Journal of International Relations 8, 375–401.

Hønneland, G., 2000. Compliance in the Barents Sea fisheries. How fishermen account for conformity with rules. Marine Policy 24, 11–19.

Horigue, V., Aliño, P.M., White, A.T., Pressey, R.L., 2012. Marine protected area networks in the Philippines: Trends and challenges for establishment and governance. Ocean & Coastal Management 64, 15-26.

Hoyos, D., 2010. The state of the art of environmental valuation with discrete choice experiments. Ecological Economics 69, 1595–1603.

IFAD, 2001. Rural poverty report - The challenge of ending rural poverty. International Fund for Agricultural Development, Rome.

Ikiara, M.M., Odink, J.G., 2000. Fishermen resistance to exit fisheries. Marine Resource Economics 14, 199–213.

ILA, 1996. More and better jobs for women: An action guide. International Labor Organization, Geneva.

Isham, J., 2001. Can Investments in Social Capital Improve Well-Being in Fishing Communities? A Theoretical Perspective for Assessing the Policy Options. CiteSeer.

Isham, J., 2002. The Effect of Social Capital on Fertiliser Adoption: Evidence from Rural Tanzania. Journal of African Economies 11, 39 –60.

Isik, M., Khanna, M., 2003. Stochastic Technology, Risk Preferences, and Adoption of Site-Specific Technologies. Am. J. Agr. Econ. 85, 305–317.

IUCN, 2005. Mnazi Bay Ruvuma Esturay Marine Park: draft general management plan. IUCN.

Jack, B.K., Kousky, C., Sims, K.R., 2008a. Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. Proceedings of the National Academy of Sciences 105, 9465–9470.

Jacoby, H.G., Skoufias, E., 1997. Risk, financial markets, and human capital in a developing country. The Review of Economic Studies 64, 311–335.

Jae, H., DelVecchio, D., 2004. Decision Making by Low-Literacy Consumers in the Presence of Point-of-Purchase Information. Journal of Consumer Affairs 38, 342–354.

Jentoft, S., Onyango, P., Mahmudul Islam, M., 2010. Freedom and poverty in the fishery commons. International Journal of the Commons 4, 345–366.

Jiddaw, N.S., 2001. Marine fisheries in Tanzania, in: Present State of Marine Science. TCMP working documents, Tanzania, pp. 64–99.

Jiddawi, N.S., Öhman, M.C., 2002. Marine fisheries in Tanzania. AMBIO: A Journal of the Human Environment 31, 518–527.

Jindal, R., Kerr, J.M., Ferraro, P.J., Swallow, B.M., 2013. Social dimensions of procurement auctions for environmental service contracts: Evaluating tradeoffs between cost-effectiveness and participation by the poor in rural Tanzania. Land Use Policy 31, 71-80.

Jindal, R., Tacconi, L., Mahanty, S., Suich, H., 2010. Livelihood impacts of payments for forest carbon services: field evidence from Mozambique. Payments for environmental services, forest conservation and climate change: livelihoods in the REDD? 185–211.

Johnson, A.E., Cinner, J.E., Hardt, M.J., Jacquet, J., McClanahan, T.R., Sanchirico, J.N., 2013. Trends, current understanding and future research priorities for artisanal coral reef fisheries research. Fish and Fisheries 14, 281-292.

Kabii, T., Horwitz, P., 2006. A review of landholder motivations and determinants for participation in conservation covenanting programmes. Environmental Conservation 33, 11–20.

Kaczan, D., Swallow, B.M., Adamowicz, W.L., 2013. Designing payments for ecosystem services (PES) program to reduce deforestation in Tanzania: An assessment of payment approaches. Ecological Economics 95, 20–30.

Kahneman, D., Knetsch, J.L., Thaler, R.H., 1990. Experimental tests of the endowment effect and the Coase theorem. Journal of political Economy 1325–1348.

Kahneman, D., Knetsch, J.L., Thaler, R.H., 1991. Anomalies: The Endowment Effect, Loss Aversion, and Status Quo Bias. The Journal of Economic Perspectives 5, 193–206.

Kelleher, G., Bleakley, C., Wells, S., 1995. A global representative system of marine protected areas. World Bank, Washington D.C.

Kellner, J.B., Tetreault, I., Gaines, S.D., Nisbet, R.M., 2007. Fishing the line near marine reserves in single and multispecies fisheries. Ecological Applications 17, 1039–1054.

Kelly, P., Huo, X., 2013. Land Retirement and Nonfarm Labor Market Participation: An Analysis of China's Sloping Land Conversion Program. World Development 48, 156–169.

Kemkes, R.J., Farley, J., Koliba, C.J., 2010. Determining when payments are an effective policy approach to ecosystem service provision. Ecological Economics 69, 2069-2074.

Kerr, J., 2002. Watershed development, environmental services, and poverty alleviation in India. World Development 30, 1387–1400.

Kerr, J., Vardhan, M., Jindal, R., 2012. Prosocial behavior and incentives: Evidence from field experiments in rural Mexico and Tanzania. Ecological Economics 73, 220–227.

Khandker, S.R., 2005. Microfinance and poverty: Evidence using panel data from Bangladesh. The World Bank Economic Review 19, 263–286.

Kiss, A., 2004. Making biodiversity conservation a land-use priority, in: Getting Biodiversity Projects to Work: Towards More Effective Conservation and Development. Columbia University Press, New York, pp. 98–123.

Kitchenham, B., Pfleeger, S.L., 2002. Principles of survey research: part 5: populations and samples. SIGSOFT Softw. Eng. Notes 27, 17–20.

Knowler, D., Bradshaw, B., 2007. Farmers' adoption of conservation agriculture: A review and synthesis of recent research. Food policy 32, 25–48.

Kocur, G., Adler, T., Hyman, W., Aunet, B. 1981. Guide to Forecasting Travel Demand with Direct Utility Assessment, Resource Policy Center, Thayer School of Engineering Dartmouth College

Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics 69, 1228–1236.

Kosoy, N., Corbera, E., Brown, K., 2008. Participation in payments for ecosystem services: Case studies from the Lacandon rainforest, Mexico. Geoforum 39, 2073–2083.

Kossoy, A., Guigon, P., 2012. State and trends of the carbon market: 2012. Washington, DC: The World Bank.

Krishna, A., 2003. Understanding, measuring and utilizing social capital: clarifying concepts and presenting a field application from India. CAPRi working paper 28. International Food Policy Research Institute., Washington DC.

Krishna, A., Uphoff, N.T., 1999. Mapping and measuring social capital: A conceptual and empirical study of collective action for conserving and developing watersheds in Rajasthan, India. World Bank, Social Development Family, Environmentally and Socially Sustainable Development Network.

Kroeger, T., Casey, F., 2007. An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecological Economics 64, 321–332.

Kuperan, K., Sutinen, J.G., 1998. Blue water crime: Deterrence, legitimacy, and compliance in fisheries. Law and Society Review 309–338.

Kurien, J., Paul, A., 2001. Social security nets for marine fisheries, Working Paper 318. Centre for Development Studies, Thiruvananthapuram.

Landell-Mills, N., 2002. Developing markets for forest environmental services: an opportunity for promoting equity while securing efficiency? Phil. Trans. R. Soc. Land. A 360, 1817–1825.

Landell-Mills, N., Porras, I., 2002. Silver bullet or fools' gold? A global review of markets for forestry environmental services and their impact on the poor. International Institute for Environment and Development, London.

Langholz, J.A., Lassoie, J.P., Lee, D., Chapman, D., 2000. Economic considerations of privately owned parks. Ecological Economics 33, 173–183.

Langpap, C., 2004. Conservation incentives programs for endangered species: an analysis of landowner participation. Land Economics 80, 375–388.

Larrazábal, A., McCall, M.K., Mwampamba, T.H., Skutsch, M., 2012. The role of community carbon monitoring for REDD+: a review of experiences. Current Opinion in Environmental Sustainability 4, 707-716.

Lau, W.W.Y., 2012. Beyond carbon: Conceptualizing payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. Ocean & Coastal Management 83, 5-14.

Lele, S., Wilshusen, P., Brockington, D., Seidler, R., Bawa, K., 2010. Beyond exclusion: alternative approaches to biodiversity conservation in the developing tropics. Current Opinion in Environmental Sustainability 2, 94–100.

Leslie, H.M., 2005. A synthesis of marine conservation planning approaches. Conservation Biology 19, 1701–1713.

Li, J., Feldman, M.W., Li, S., Daily, G.C., 2011. Rural household income and inequality under the Sloping Land Conversion Program in western China. Proceedings of the National Academy of Sciences 108, 7721–7726.

Liang, Y., Li, S., Feldman, M.W., Daily, G.C., 2012. Does household composition matter? The impact of the Grain for Green Program on rural livelihoods in China. Ecological Economics 75, 152–160.

Lindahl, O., Kollberg, S., 2009. Can the EU agri-environmental aid program be extended into the coastal zone to combat eutrophication? Hydrobiologia 629, 59–64.

Lopa, D., Mwanyoka, I., Jambiya, G., Massoud, T., Harrison, P., Ellis-Jones, M., Blomley, T., Leimona, B., van Noordwijk, M., Burgess, N.D., 2012. Towards operational payments for water ecosystem services in Tanzania: a case study from the Uluguru Mountains. Oryx 46, 34–44.

Louviere, J.J., Hensher, D.A., Swait, J.D., 2000. Stated choice methods: analysis and applications. Cambridge Univ Pr.

Lowe, T.D., Lorenzoni, I., 2007. Danger is all around: Eliciting expert perceptions for managing climate change through a mental models approach. Global Environmental Change 17, 131–146.

Lundquist, C.J., Granek, E.F., 2005. Strategies for Successful Marine Conservation: Integrating Socioeconomic, Political, and Scientific Factors. Conservation Biology 19, 1771.

Lunn, K.E., Dearden, P., 2006. Monitoring small-scale marine fisheries: An example from Thailand's Ko Chang archipelago. Fisheries Research 77, 60–71.

Lyster, R., 2011. REDD+, transparency, participation and resource rights: the role of law. Environmental science & policy 14, 118–126.

MA, 2005. Ecosystems and human well-being. Washington, DC: Island Press.

Ma, S., Swinton, S.M., Lupi, F., Jolejole, C.B., 2010. Why Farmers Opt Not to Enroll in Payment-for-Environmental-Services Programs. Selected Paper prepared for presentation at the Agricultural & Applied Economics Association's 2010 AAEA, CAES, & WAEA Joint Annual Meeting in Denver, July 25-July 27, 2010.

MacKenzie, I.A., Ohndorf, M., Palmer, C., 2012. Enforcement-proof contracts with moral hazard in precaution: ensuring "permanence"in carbon sequestration. Oxford economic papers 64, 350–374.

Madsen, B., Carroll, N., Kandy, D., Allen, G.R., 2011. Update: State of Biodiversity Markets. Forest Trends, Washington, DC.

Mahanty, S., Suich, H., Tacconi, L., 2013. Access and benefits in payments for environmental services and implications for REDD+: Lessons from seven PES schemes. Land Use Policy 31, 38–47.

Malleret, D., 2004. A socio-economic baseline assessment of the Mnazi Bay Ruvuma Estuary Marine Park. IUCN EARO, Nairobi, Kenya.

Malleret, D., Simbua, J., 2004. The occupational structure of the Mnazi Bay Ruvuma Estuary Marine Park communities. IUCN EARO, Nairobi, Kenya.

Mandel, J.T., Donlan, C.J., Wilcox, C., Cudney-Bueno, R., Pascoe, S., Tulchin, D., 2009. Debt investment as a tool for value transfer in biodiversity conservation. Conservation Letters 2, 233–239.

Mariola, M.J., 2012. Farmers, trust, and the market solution to water pollution: The role of social embeddedness in water quality trading. Journal of Rural Studies 28, 577-589.

Masalu, D.C.., 2000. Coastal and marine resource use conflicts and sustainable development in Tanzania. Ocean & Coastal Management 43, 475–494.

Mascia, M.B., 2003. The human dimension of coral reef marine protected areas: recent social science research and its policy implications. Conservation Biology 17, 630–632.

Mathew, S., 2001. Small-scaled fisheries perspectives on an ecosystem-based approach to fisheries management, in: Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem.

Matuschke, I., Qaim, M., 2009. The impact of social networks on hybrid seed adoption in India. Agricultural Economics 40, 493–505.

McAfee, K., Shapiro, E.N., 2010. Payments for ecosystem services in Mexico: nature, neoliberalism, social movements, and the state. Annals of the Association of American Geographers 100, 579–599.

McClanahan, T., 1999. Is there a future for coral reef parks in poor tropical countries? Coral Reefs 18, 321–325.

McClanahan, T.R., 2010. Effects of fisheries closures and gear restrictions on fishing income in a Kenyan coral reef. Conservation Biology 24, 1519-1528.

McClanahan, T., Davis, J., Maina, J., 2005. Factors infuencing resource users and manager's perceptions towards marine protected area management in Kenya. Foundations for Environmental Conservation 32, 42–49.

McClanahan, T.R., Cinner, J., Kamukuru, A.T., Abunge, C., Ndagala, J., 2009. Management preferences, perceived benefits and conflicts among resource users and managers in the Mafia Island Marine Park, Tanzania. Environmental Conservation 35, 340–350.

McClanahan, T.R., Mangi, S.C., 2004. Gear-based management of a tropical artisanal fishery based on species selectivity and capture size. Fisheries Management & Ecology 11, 51–60.

McCook, L.J., Ayling, T., Cappo, M., Choat, J.H., Evans, R.D., De Freitas, D.M., Heupel, M., Hughes, T.P., Jones, G.P., Mapstone, B., others, 2010. Adaptive management of the Great Barrier Reef: A globally significant demonstration of the benefits of networks of marine reserves. Proceedings of the National Academy of Sciences 107, 18278–18285.

McFadden, D., 1974. Conditional logit analysis of qualitative choice behavior, in: Frontiers in Econometrics. Academic Press, New York, pp. 105–142.

McGoodwin, J.R., 2001. Understanding the cultures of fishing communities: a key to fisheries management and food security. Food & Agriculture Organisation, Rome.

Meinzen-Dick, R., DiGregorio, M., McCarthy, N., 2004. Methods for studying collective action in rural development. Agricultural Systems 82, 197–214.

Mendola, M., 2005. Farm Households Production Theories: A Review of "Institutional" and "Behavioural" Responses (SSRN Scholarly Paper No. ID 661021). Social Science Research Network, Rochester, NY.

Mercer, D.E., 2004. Adoption of agroforestry innovations in the tropics: A review. Agroforestry Systems 61-62, 311–328.

Meyerhoff, J., Liebe, U., 2009. Status Quo Effect in Choice Experiments: Empirical Evidence on Attitudes and Choice Task Complexity. Land Economics 85, 515–528.

Mills, D., Béné, C., Ovie, S., Tafida, A., Sinaba, F., Kodio, A., Russell, A., Andrew, N., Morand, P., Lemoalle, J., 2009. Vulnerability in African small-scale fishing communities. Journal of International Development 23, 308-313.

Milne, S., Adams, B., 2012. Market Masquerades: Uncovering the Politics of Community-level Payments for Environmental Services in Cambodia. Development and Change 43, 133–158.

Ministry of Finance and Economic Affairs, 2010. Eradication and Economic Empowerment Division, United Republic of Tanzania: Tanzania Gender Indicators Booklet. Dar es Salaam, Tanzania.

Miranda, M., Porras, I.., Moreno, M., 2003. The social impacts of payments for environmental services in Cosat Rica. A quantitative field survey and analysis of the Virilla watershed, Markets for Environmental Services. International Institute for Environmental and Development, London.

Mitchell, R.C., Carson, R.T., 1993. Using surveys to value public goods: the contingent valuation method. Resources for the Future.

Miteva, D.A., Pattanayak, S.K., Ferraro, P.J., 2012. Evaluation of biodiversity policy instruments: what works and what doesn't? Oxford Review of Economic Policy 28, 69–92.

Moberg, F., Rönnbäck, P., 2003. Ecosystem services of the tropical seascape: interactions, substitutions and restoration. Ocean & Coastal Management 46, 27–46.

Mogas, J., Riera, P., Bennett, J., 2006. A comparison of contingent valuation and choice modelling with second-order interactions. Journal of Forest Economics 12, 5–30.

Mohammed, E.Y., 2012. Payments for coastal and marine ecosystem services: prospects and principles, IIED Briefing. IIED, London.

Molyneux, M., 2002. Gender and the silences of social capital: lessons from Latin America. Development and Change 33, 167–188.

Moore, R.H., Parker, J.S., Weaver, M., 2008. Agricultural sustainability, water pollution, and governmental regulations: Lessons from the Sugar Creek farmers in Ohio. Culture & Agriculture 30, 3–16.

Mora, C., Andrefouet, S., Costello, M.J., Kranenburg, C., Rollo, A., Veron, J., Gaston, K.J. & Myers, R.A., 2006. How protected are coral reefs? Response. Science 314, 757-760.

Morgan, M.G., Henrion, M., 1992. Uncertainty: a guide to dealing with uncertainty in quantitative risk and policy analysis. Cambridge Univ Pr.

Moser, C., 2008. Moser, C. & Dani, A. (eds), Assets and Livelihoods: a framework for asset-based social policy, in: Assets, Livelihoods, and Social Policy. World Bank Publications.

Moser, C.O.N., 1998. The asset vulnerability framework: reassessing urban poverty reduction strategies. World development 26, 1–19.

Moustakas, C.E., 1994. Phenomenological research methods. Sage Publications, Inc.

Muallil, R.N., Geronimo, R.C., Cleland, D., Cabral, R.B., Doctor, M.V., Cruz-Trinidad, A., Aliño, P.M., 2011. Willingness to exit the artisanal fishery as a response to scenarios of declining catch or increasing monetary incentives. Fisheries Research 111, 74–81.

Mullan, K., Kontoleon, A., 2009. Participation in Payments for Ecosystem Services programmes in developing countries: The Chinese Sloping Land Conversion Programme. Environmental Economy and Policy Research Working Papers 42.2009. University of Cambridge, Department of Land Economics.

Muller, J., Albers, H.J., 2004. Enforcement, payments, and development projects near protected areas: how the market setting determines what works where. Resource and Energy Economics 26, 185–204.

Muñoz-Pina. C., 2009, Instituto Nacional de Ecología, Ministry of the Environment, Mexico., Instituto Nacional de Ecología. *Pers comm*.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69, 1202–1208.

Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., Aguilar, B., Agarwal, B., Corbera, E., de Blas, D.E., Farley, J., Froger, G., 2013. Payments for ecosystem services and the fatal attraction of win-win solutions. Conservation Letters 6, 274-279.

Murray, B.C., Pendleton, L., Jenkins, W.A., Sifleet, S., 2011. Green payments for blue carbon: Economic incentives for protecting threatened coastal habitats (No. NI R 11-04), Nicholas Institute for Environmental Policy Solutions. Nicholas Institute for Environmental Policy Solutions, Duke University.

Murray, B.C., Sohngen, B., 2004. Leakage in land use, land use change, and forest sector projects: A synthesis of issues and estimation approaches. Working paper, RTI International. Center for Regulatory Economics and Policy Research, Research Triangle Park, NC.

Musyoki, J.K., Mugwe, J., Mutundu, K., Muchiri, M., 2013. Determinants of Household Decision to Join Community Forest Associations: A Case Study of Kenya. ISRN Forestry 2013, 1-10.

Mwangi, E., Meinzen-Dick, R., Sun, Y., 2011. Gender and sustainable forest management in East Africa and Latin America. Ecology and society 16, 17.

Naef, M., Schupp, J., 2009. Measuring trust: experiments and surveys in contrast and combination. Deutsches Institut für Wirtschaftsforschung, Berlin.

Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H., Rouget, M., 2006. Integrating economic costs into conservation planning. Trends in Ecology & Evolution 21, 681–687.

Narayan, D., Pritchett, L., 1997. Cents and sociability : household income and social capital in rural Tanzania. The World Bank.

Narloch, U., Pascual, U., Drucker, A.G., 2013. How to achieve fairness in payments for ecosystem services? Insights from agrobiodiversity conservation auctions. Land Use Policy 35, 107–118.

Nathan, D., Apu, N.A., 1998. Women's Independent Access to Productive Resources: Fish Ponds in the Oxbow Lakes Project, Bangladesh. Gender, Technology and Development 2, 397–413.

National Bureau of Statistics (NBS), N.B.S., 2002. National Household Budget Survey 2000-2001. National Bureau of Statistics, Dar es Salaam, Tanzania.

National Bureau of Statistics (NBS), N.B.S., 2007. Household Budget Survey 2007. National Bureau of Statistics, Dar es Salaam, Tanzania.

Neill, H.R., Cummings, R.G., Ganderton, P.T., Harrison, G.W., McGuckin, T., 1994. Hypothetical surveys and real economic commitments. Land Economics 70, 145–154.

Neill, S.P., Lee, D.R., 2001. Explaining the adoption and disadoption of sustainable agriculture: the case of cover crops in northern Honduras. Economic Development and Cultural Change 49, 793–820.

Nellemann, C., Corcoran, E., Duarte, C.., Valdez, I., DeYoung, C., Fonseca, L., Grimsditch, G., 2009. Blue carbon: the role of healthy oceans in binding carbon: a rapid response assessment. United Nations Environmental Programme, GRID-Arendal.

Nelson, F., Foley, C., Foley, L.S., Leposo, A., Loure, E., Peterson, D., Peterson, M., Peterson, T., Sachedina, H., Williams, A., 2010. Payments for Ecosystem Services as a Framework for Community-Based Conservation in Northern Tanzania. Conservation Biology 24, 78–85.

Newburn, D., Reed, S., Berck, P., Merenlender, A., 2005. Economics and Land-Use Change in Prioritizing Private Land Conservation. Conservation Biology 19, 1411–1420.

Newton, K., 2007. Social and political trust. Oxford Handbook of Political Behavior 342. DOI: 10.1093/oxfordhb/9780199270125.003.0018

Newton, P., Nichols, E.S., Endo, W., Peres, C.A., 2012. Consequences of actor level livelihood heterogeneity for additionality in a tropical forest payment for environmental services programme with an undifferentiated reward structure. Global Environmental Change 22, 127–136.

Nguyen, C.H., 2006. Determinants of credit participation and its impact on household consumption: evidence from rural Vietnam, in: Paper Presented at the 3rd Leicester PhD Conference on Economics. p. 21.

Nkamleu, G.B., Manyong, V.M., 2005. Factors affecting the adoption of agroforestry practices by farmers in Cameroon. Small-scale Forestry 4, 135–148.

Nobre, A.M., 2011. Scientific approaches to address challenges in coastal management. Mar Ecol Prog Ser 434, 279–289.

Norton, E.C., Wang, H. & Ai. C., 2004. Computing interaction effects and standard errors in logit and probit models. *Stata Journal,* 4, pp.154-167.

Nuggehalli, R.K., Prokopy, L.S., 2009. Motivating factors and facilitating conditions explaining women's participation in co-management of Sri Lankan forests. Forest policy and economics 11, 288–293.

Nyangena, W., 2008. Social determinants of soil and water conservation in rural Kenya. Environ Dev Sustain 10, 745–767.

O'Neill, S.J., Osborn, T.J., Hulme, M., Lorenzoni, I., Watkinson, A.R., 2008. Using expert knowledge to assess uncertainties in future polar bear populations under climate change. Journal of Applied Ecology 45, 1649–1659.

OECD, 2001. Poverty-Environment Gender Linkages. Organisation for Economic Co-operation and Development, Paris.

Okoye, C.U., 1998. Comparative analysis of factors in the adoption of traditional and recommended soil erosion control practices in Nigeria. Soil and Tillage Research 45, 251–263.

Olale, E., Henson, S., Cranfield, J., 2010. Determinants of Income Diversification among Fishing Communities in Western Kenya, in: Agricultural & Applied Economics Association. Presented at the AAEA, CAES, & WAEA Joint Annual Meeting, Denver, Colorado.

Olli, E., Grendstad, G., Wollebaek, D., 2001. Correlates of Environmental Behaviors Bringing Back Social Context. Environment and Behavior 33, 181–208.

Ortiz Malavasi, E., Sage Mora, F.F., Borge Carvajal, C., 2003. Impacto del programa de pago de servicios ambientales en Costa Rica como medio de recucción de la pobreza en los medios rurales. Unidad Regional de Asistencia Técnica (RUTA), San José.

Ostrom, E., 1999. Social capital: fad or a fundamental concept?, in: Social Capital: A Multifaceted Perspective, IBRD. The World Bank, Washington D.C.

Ostrom, E., Ahn, T.K., 2001. A social science perspective on social capital: social capital and collective action. Presented at the European Research Conference on Social Capital: Interdisciplinary Perpsectives, Exeter, UK.

Ostrom, E., 1990. Governing the commons: The evolution of institutions for collective action. Cambridge Univ Pr.

Ostrom, E., 2010. Analyzing collective action. Agricultural Economics 41, 155–166.

Oxoby, R., 2009. Understanding social inclusion, social cohesion, and social capital. International Journal of Social Economics 36, 1133–1152.

Padmanabhan, M.A., 2008. Collective action in agrobiodiversity management: gendered rules of reputation, trust and reciprocity in Kerala, India. Journal of International Development 20, 83–97.

Pagiola, S., 2008. Can payments for environmental services help protect coastal and marine areas? Environment Matters 14–17.

Pagiola, S., Agostini, P., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., Ramirez, E., Rosales, M., Ruiz, J.P., 2004. Paying for biodiversity conservation services in agricultural landscapes. The International Bank for Reconstruction and Development, The World Bank, Washington, D.C.

Pagiola, S., Arcenas, A., Platais, G., 2005. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development 33, 237–253.

Pagiola, S., Platais, G., 2007. Payments for Environmental Services: From Theory to Practice. World Bank, Washington, DC.

Pagiola, S., Rios, A.R., Arcenas, A., 2008. Can the poor participate in payments for environmental services? Lessons from the Silvopastoral Project in Nicaragua. Environment and Development Economics 13, 299–325.

Palmer, C., 2011. Property rights and liability for deforestation under REDD+: Implications for "permanence" in policy design. Ecological Economics 70, 571–576.

Pandolfelli, L., Meinzen-Dick, R., Dohrn, S., 2008. Gender and collective action: motivations, effectiveness and impact. Journal of International Development 20, 1–11.

Parker, J.S., Moore, R., Weaver, M., 2009. Developing participatory models of watershed management in the Sugar Creek watershed (Ohio, USA). Water Alternatives 2, 82–100.

Pascual, U., Muradian, R., Rodríguez, L.C., Duraiappah, A., 2010. Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics 69, 1237–1244.

Pattanayak, S.K., Evan Mercer, D., Sills, E., Yang, J.C., 2003. Taking stock of agroforestry adoption studies. Agroforestry Systems 57, 173–186.

Pattanayak, S.K., Wunder, S., Ferraro, P.J., 2010. Show me the money: do payments supply environmental services in developing countries? Review of Environmental Economics and Policy 4, 254.

Pauly, D., 2006. Major trends in small-scale marine fisheries, with emphasis on developing countries, and some implications for the social sciences. Maritime Studies (MAST) 4, 7–22.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumalia, U.R., Walters, C.J., Watson, R., Zeller, D., 2002. Towards sustainability in world fisheries. Nature 418, 689–695.

Pearlman, S., 2012. Too Vulnerable for Microfinance? Risk and Vulnerability as Determinants of Microfinance Selection in Lima. Journal of Development Studies 48, 1342–1359.

Pedroni, L., Dutschke, M., Streck, C., Porrúa, M.E., 2009. Creating incentives for avoiding further deforestation: the nested approach. Climate Policy 9, 207–220.

Perrot-Maitre, D., 2006. The Vittel payments for ecosystem services: a "perfect" PES case. IIED and DFID. Available at www.iied.org/NR/forestry/documents/Vittelpaymentsforecosystemservices.pdf (Accessed 8 May 2008).

Pet-Soede, C., Van Densen, W.L.T., Hiddink, J.G., Kuyl, S., Machiels, M.A.M., 2001. Can fishermen allocate their fishing effort in space and time on the basis of their catch rates? An example from Spermonde Archipelago, SW Sulawesi, Indonesia. Fisheries Management and Ecology 8, 15–36.

Peters-Stanley, M., Hamilton, K., Marcello, T., Sjardin, M., 2011. Back to the Future: State of the Voluntary Carbon Markets 2011. Ecosystem Marketplace adn Bloomberg New Energy Finance, London.

Peterson, A.M., Stead, S.M., 2011. Rule breaking and livelihood options in marine protected areas. Environmental Conservation 38, 342–352.

Petheram, L., Campbell, B.M., 2010. Listening to locals on payments for environmental services. Journal of Environmental Management 91, 1139–1149.

Petter Johnsen, J., Eliasen, S., 2011. Solving complex fisheries management problems: What the EU can learn from the Nordic experiences of reduction of discards. Marine Policy 35, 130–139.

Pfaff, A., Robalino, J.A., Sanchez-Azofeifa, G.A., 2008. Payments for environmental services: empirical analysis for Costa Rica., Working Papers Series SAN08-05. Terry Sanford Institute of Public Policy, Duke University, Durham.

Phelps, J., Webb, E.L., Agrawal, A., 2010. Does REDD+ threaten to recentralize forest governance. Science 328, 312–313.

Phelps, J., Webb, E.L., Koh, L.P., 2011. Risky business: an uncertain future for biodiversity conservation finance through REDD+. Conservation Letters 4, 88–94.

Pikitch, E.K., Santora, C., Babcock, E.A., Bakun, A., Bonfil, R., Conover, D.O., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E.D., Link, J., Livingston, P.A.,

Mangel, M., McAllister, M.K., Pope, J., Sainsbury, K.J., 2004. Ecosystem-Based Fishery Management. Science 305, 346–347.

Pokharel, B., Carter, J., Parajuli, R.R., Byrne, S., Gurung, B.D., 2009. Community forestry in Nepal as a means of empowering people living in poverty: An assessment of its social, economic, and environmental sustainability, in: A Paper Presented at the International Community Forestry Workshop Held in Pokhara, Nepal during September. pp. 15–18.

Polanyi, K., 1944. The Great Transformation. Beacon Press, Boston.

Pollnac, R., Crawford, B.R., Gorospe, M.L.G., 2001. Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. Ocean & Coastal Management 44, 683–710.

Pollnac, R., 1991. Social and cultural characteristics in small-scale fishery development, in: M.M. Cernea, Putting People First. Sociological Variable in Rural Development. World Bank (by) Oxford University Press, New York.

Polman, N.B.P., Slangen, L.H.G., 2008. Institutional design of agri-environmental contracts in the European Union: the role of trust and social capital. NJAS - Wageningen Journal of Life Sciences 55, 413–430.

Pomeroy, Katon, B.M., Harkes, I., 2001. Conditions affecting the success of fisheries co-management: lessons from Asia. Marine Policy 25, 197–208.

Pomeroy, R.S., 2001. Devolution and fisheries co-management. Collective Action, Property Rights and Devolution of Natural Resource Management–Exchange of Knowledge and Implications for Policy. Zentralstelle fur Ernahrung und Landwirtschaft, Feldafing, Germany.

Pomeroy, R.S., Oracion, E.G., Pollnac, R.B., Caballes, D.A., 2005. Perceived economic factors influencing the sustainability of integrated coastal management projects in the Philippines. Ocean & coastal management 48, 360–377.

Porter, M., Mbezi, R.G., 2010. From Hand to Mouth: Fishery Projects, Women, Men and Household Poverty. Canadian Journal of Development Studies/Revue canadienne d'études du développement 31, 381–400.

Poteete, A.R., Ostrom, E., 2004. Heterogeneity, group size and collective action: The role of institutions in forest management. Development and change 35, 435–461.

Pradhan, N.C., Leung, P., 2004. Modeling entry, stay, and exit decisions of the longline fishers in Hawaii. Marine Policy 28, 311–324.

Prager, K., Prazan, J., Penov, I., 2012. Soil Conservation in Transition Countries: the Role of Institutions. Environmental Policy and Governance 22, 55–73.

Pretty, J., 2003. Social capital and the collective management of resources. Science 302, 1912.

Pretty, J., 2008. Investments in collective capacity and social capital, in: Conserving Land, Protecting Water. CAB International, Oxfordshire, p. 235.

Pretty, J., Ward, H., 2001. Social capital and the environment. World development 29, 209–227.

Putnam, R.D., 1995. Bowling alone: America's declining social capital. Journal of democracy 6, 65–78.

Qin, P., Carlsson, F., Xu, J., 2011. Forest Tenure Reform in China: A Choice Experiment on Farmers' Property Rights Preferences. Land Economics 87, 473–487.

Raymond, L., 2006. Cooperation without Trust: Overcoming Collective Action Barriers to Endangered Species Protection. Policy Studies Journal 34, 37–57.

Reardon, T., Berdegué, J., Barrett, C.B., Stamoulis, K., 2007. Household income diversification into rural nonfarm activities. Transforming the Rural Nonfarm Economy. Opportunities and Threats in the Developing World. Johns Hopkins University Press, Baltimore.

Redford, K., Adams, W., 2009. Payment for Ecosystem Services and the Challenge of Saving Nature. Conservation Biology 23, 785–787.

Redko, A., 2013. Community-Based Conditional Cash Transfer Impact Evaluation 2009, Baseline Survey. Tanzania National Data Archieve (TNADA).

Resurreccion, B.P., 2006. Rules, Roles and Rights: Gender, Participation and Community Fisheries Management in Cambodia's Tonle Sap Region. International Journal of Water Resources Development 22, 433–447.

Richards, D., 1996. Elite interviewing: approaches and pitfalls. Politics 16, 199–204.

Roberts, C.M., Bohnsack, J.A., Gell, F., Hawkins, J.P., Goodridge, R., 2001. Effects of marine reserves on adjacent fisheries. science 294, 1920–1923.

Robertson, N., Wunder, S., 2005. Fresh tracks in the forest: assessing incipient payments for environmental services initiatives in Bolivia. CIFOR, Bogor, Indonesia.

Robinson, E.J.Z., Albers, H.J., Kirama, S.L., 2012. The role of incentives for sustainable implementation of marine protected areas: an example from Tanzania, Environment for Development Discussion Paper Series. Environment for Development.

Rosenzweig, M.R., 1988. Risk, implicit contracts and the family in rural areas of low-income countries. The Economic Journal 1148–1170.

Russ, G.R., Alcala, A.C., Maypa, A.P., 2003. Spillover from marine reserves: the case of Naso vlamingii at Apo Island, the Philippines. Marine Ecology Progress Series 264, 15–20.

Ruto, E., Garrod, G., 2009. Investigating farmers' preferences for the design of agri-environment schemes: a choice experiment approach. Journal of Environmental Planning and Management 52, 631–647.

Salzman, J., 2005. The promise and perils of payments for ecosystem services. International Journal of Innovation and Sustainable Development 1, 5–20.

Samoilys, M., 2010. *Specification of trial conservation activities in the Mnazi Bay - Ruvuma Estuary Marine Park.* Report for CARE East African Regional Office, 15pp.

Samuelson, W., Zeckhauser, R., 1988. Status quo bias in decision making. Journal of Risk and Uncertainty 1, 7–59.

Schläpfer, F., 2006. Survey protocol and income effects in the contingent valuation of public goods: A meta-analysis. Ecological Economics 57, 415–429.

Sedjo, R.A., Marland, G., 2003. Inter-trading permanent emissions credits and rented temporary carbon emissions offsets: some issues and alternatives. Climate Policy 3, 435–444.

Sekhar, N.U., 2007. Social capital and fisheries management: the case of Chilika Lake in India. Environmental Management 39, 497–505.

Sesabo, J.K., Lang, H. & Tol, R.S., 2006. Perceived Attitude and Marine Protected Areas (MPAs) establishment: Why households' characteristics matters in Coastal resources conservation initiatives in Tanzania. *Working Paper FNU-99*. Hamburg University and Centre for Marine Atmospheric Sciences.

Sesabo, J.K. & Tol, R.S., 2005. *Factors affecting Income Strategies among households in Tanzanian Coastal Villages: Implications for Development-conservation initiatives*, Research unit Sustainability and Global Change, *Working Paper FNU-70*. Hamburg University and Centre for Marine Atmospheric Sciences.

Shao, F., Mlay, E., Mushi, V., 2003. Review for marine fisheries for Kenya. Report for DFID study: understanding fisheries associated livelihoods and the constraints to their development in Kenya and Tanzania (Project R8196).

Shelley, B.G., 2011. What should we call instruments commonly known as payments for environmental services? A review of the literature and a proposal. Annals of the New York Academy of Sciences 1219, 209–225.

Shete, M., Garcia, R.J., 2011. Agricultural credit market participation in Finoteselam town, Ethiopia. Journal of Agribusiness in Developing and Emerging Economies 1, 55–74.

Sievanen, L., Crawford, B., Pollnac, R., Lowe, C., 2005. Weeding through assumptions of livelihood approached in ICM: seaweed farming in the Philippines and Indonesia. Ocean & Coastal Management 48, 297–313.

Silva, P., 2006. Exploring the linkages between poverty, marine protected area management, and the use of destructive fishing gear in Tanzania. Policy Research Working Paper 383, Washington DC: The World Bank.

Skutsch, M., Vickers, B., Georgiadou, Y., McCall, M., 2011. Alternative models for carbon payments to communities under REDD+: A comparison using the Polis model of actor inducements. Environmental Science & Policy 14, 140–151.

Smale, M., Bellon, M.R., Gomez, J.A.A., 2001. Maize Diversity, Variety Attributes, and Farmers' Choices in Southeastern Guanajuato, Mexico*. Economic Development and Cultural Change 50, 201–225.

Smith, T.F., Alcock, D., Thomsen, D.C., Chuenpagdee, R., 2006. Improving the quality of life in coastal areas and future directions for the Asia-Pacific Region. Coastal Management 34, 235–250.

Sobo, F.S., 2012. Community Participation in Fisheries Management in Tanzania. In: Visible Possibilities: The Economics of Sustainable Fisheries, Aquaculture and Seafood Trade: Proceedings of the Sixteenth Biennial Conference of the International Institute of Fisheries Economics and Trade, July 16-20, Dar es Salaam, Tanzania. Edited by Ann L. Shriver. International Institute of Fisheries Economics and Trade (IIFET), Corvallis, 2012.

Sommerville, M., Jones, J.P.G., Rahajaharison, M., Milner-Gulland, E.J., 2010a. The role of fairness and benefit distribution in community-based Payment for Environmental Services interventions: A case study from Menabe, Madagascar. Ecological Economics 69, 1262–1271.

Sommerville, M., Milner-Gulland, E., Rahajaharison, M. & Jones, J.P.G., 2010b. Impact of a Community-Based Payment for Environmental Services Intervention on Forest Use in Menabe, Madagascar. Conservation biology 24, 1488–1498.

Sommerville, M.M., Jones, J.P.G., Milner-Gulland, E., 2009. A revised conceptual framework for payments for environmental services. Ecology and Society 14, 34.

Sommerville, M.M., Milner-Gulland, E.J. & Jones, J.P.G., 2011. The challenge of monitoring biodiversity in payment for environmental service interventions. Biological Conservation 144, 2832-2841.

Southgate, D., Haab, T., Lundine, J. & Rodríguez, F., 2010. Payments for environmental services and rural livelihoods in Ecuador and Guatemala. Environment and Development Economics, 15, 21-37

Standish, S., 2006. Investing in tomorrow's markets. Environmental Finance March, 24–25.

Staples, D., Satia, B., Gardiner, P., 2004. A research agenda for small-scale fisheries, RAP Publication. FAO Fisheries and Aquaculture Department, Food and Agriculture Organisation of the United Nations Regional Office for the Asia and the Pacific, Bangkok, Thailand.

Stern, M.J., 2008. Coercion, voluntary compliance and protest: the role of trust and legitimacy in combating local opposition to protected areas. Environmental Conservation 35, 200–210.

Sturmheit, P., 1990. Agroforestry and soil conservation needs of smallholders in Southern Zambia. Agroforest Syst 10, 265–289.

Sultana, P., Thompson, P., 2008. Gender and local floodplain management institutions: a case study from Bangladesh. Journal of International Development 20, 53–68.

Swallow, B.M., Kallesoe, M.F., Iftikhar, U.A., van Noordwijk, M., Bracer, C., Scherr, S.J., Raju, K.V., Poats, S.V., Duraiappah, A.K., Ochieng, B.O., Mallee, H. & Rumley, R., 2009. Compensation and rewards for environmental services in the developing world: framing pan-tropical analysis and comparison. Ecology & Society 14, 26.

Swinton, S.M., Quiroz, R., 2003. Poverty and the deterioration of natural soil capital in the Peruvian Altiplano. Environment, Development and Sustainability 5, 477–490.

Sze Choo, P., Nowak, B.S., Kusakabe, K., Williams, M.J., 2008. Guest Editorial: Gender and Fisheries. Development 51, 176–179.

Tacconi, L., 2012. Redefining payments for environmental services. Ecological Economics 73, 29–36.

Tanzania Social Action Fund, n.d. CB-CCT results: community based conditional cash transfers in Tanzania: results from a randomized trial. Accessed 01/01/2014 Available from: http://www.tasaf.org/index.php?option=com_content&view=article&id=136:cb-cct-results&catid=41:quick-links

Tanzanian Fisheries Division, 2005. National Guidelines for Beach Management Units.

Teh, L., Cheung, W.W.L., Cornish, A., Chu, C., Sumaila, U.R., 2008. A survey of alternative livelihood options for Hong Kong's fishers. International Journal of Social Economics 35, 380–395.

The World Bank, undated. Measuring the dimensions of social capital. Accessed 12/06/2013 Available at: http://web.worldbank.org/WBSITE/EXTERNAL/TOPICS/EXTSOCIALDEVELOPMENT/EXTTSOCIALCAPITAL/0,,contentMDK:20305939~menuPK:418220~pagePK:148956~piPK:216618~theSitePK:401015~isCURL:Y,00.html

The World Bank, Food and Agricultural Organization, International Fund for Agricultural Development, 2009. Module 13: Gender in Fisheries and Aquaculture, in: Gender in Agriculture. The International Bank for Reconstruction and Development / World Bank, Washington DC.

Tindall, C., Holvoet, K., 2008. From the Lake to the Plate: Assessing gender vulnerabilities throughout the fisheries chain. Development 51, 205–211.

Tobey, J., Torell, E., 2006. Coastal poverty and MPA management in mainland Tanzania and Zanzibar. Ocean & Coastal Management 49, 834–854.

Togba, E.L., 2012. Microfinance, Social Capital and Households Access to Credit: Evidence from Coŧe d'Ivoire. Structural Change and Economic Dynamics 23, 473-486.

Townsend, R.M., 1994. Risk and insurance in village India. Econometrica: Journal of the Econometric Society 62, 539–591.

Townsley, P., 1998. Livelihoods and aquatic resource management., in: In: Carney D, Editor. Sustainable Rural Livelihoods: What Contribution Can We Make? Department of International Development, London, UK, pp. 139–54.

Traore, N., Landry, R., Amara, N., 1998. On-Farm Adoption of Conservation Practices: The Role of Farm and Farmer Characteristics, Perceptions, and Health Hazards. Land Economics 74, 114.

Travers, H., Clements, T., Keane, A., Milner-Gulland, E.J., 2011. Incentives for cooperation: the effects of institutional controls on common pool resource extraction in Cambodia. Ecological Economics 71, 151–161.

Tschakert, P., 2007. Environmental services and poverty reduction: options for smallholders in the Sahel. Agricultural Systems 94, 75–86.

Turpie, J.K., Marais, C., Blignaut, J.N., 2008. The working for water programme: Evolution of a payments for a ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecological Economics 65, 788–798.

Uchida, E., Rozelle, S., Xu, J., 2009. Conservation Payments, Liquidity Constraints, and Off-Farm Labor: Impact of the Grain-for-Green Program on Rural Households in China. Am. J. Agr. Econ. 91, 70–86.

Uchida, E., Xu, J., Xu, Z., Rozelle, S., 2007. Are the poor benefiting from China's land conservation program? Environment and Development Economics 12, 593–620.

UNDP, 1995. Human Development Report 1995: Gender and Human Development Technical Report. United Nations Development Program, New York.

UNDP, 2002. UNITED NATIONS DEVELOPMENT PROGRAMME PROJECT DOCUMENT: Development of Mnazi Bay-Ruvuma Estuary Marine Park. UNDP, Washington DC.

UNDP, 2006. Human Development Report 2006: Beyond scarcity: Power, poverty and the global water crisis. United Nations Development Program, Washington D.C.

UNDP, 2011. International Human development Indicators: Tanzania (United Republic of) Available at: http://hdrstats.undp.org/en/countries/profiles/TZA.html [Accessed September 4, 2012].

UNEP, 2006. Marine and coastal ecosystems and human well-being: a synthesis report based on the findings of the Millennium Ecosystem Assessment.

UNFCCC, 2005. Report of the 11th Conference of the Parties serving as the meeting of the Parties to the Kyoto Protocol on its first session, held at Montreal from 28th Novermber to 10th December 2005 - addendum - part two: action taken by the Conference of the Parties serving as the meeting of the Parties to the Kyoto Protocal at its first session. http://unfccc.int/resource/docs/2005/cmp1/eng/08a01.pdf.

Uphoff, N., 2000. Understanding social capital: learning from the analysis and experience of participation. Social Capital: A multifaceted perspective 215–249.

Van der Horst, D., 2011. Adoption of payments for ecosystem services: An application of the Hägerstrand model. Applied Geography 31, 668–676.

Van Dusen, M., Taylor, J.E., 2005. Missing markets and crop diversity: evidence from Mexico. Environment and Development Economics 10, 513–531.

Van Hecken, G., Bastiaensen, J., 2010. Payments for ecosystem services: justified or not? A political view. Environmental Science & Policy 13, 785–792.

Van Noordwijk, M., Leimona, B., Emerton, L., Tomich, T.P., Velarde, S.J., Kallesoe, M., Sekher, M., Swallow, B., 2007. Criteria and indicators for environmental service compensation and reward mechanisms: realistic, voluntary, conditional and pro-poor: CES Scoping Study Issue Paper no. 2. ICRAF Working Paper no. 37. World Agroforestry Centre, Nairobi, Kenya.

Vanslembrouck, I., Huylenbroeck, G., Verbeke, W., 2005. Determinants of the Willingness of Belgian Farmers to Participate in Agri-environmental Measures. Journal of agricultural economics 53, 489–511.

Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics 69, 1245–1252.

Wagner, G.M., Akwilapo, F.D., Mrosso, S., Ulomi, S., Masinde, R., 2004. Assessment of Marine Biodiversity, Ecosystem Health, and Resource Status in Mangrove Forests in Mnazi Bay Ruvuma Estuary Marine Park, viii+ 106pp.

Wakefield, S.E.L., Poland, B., 2005. Family, friend or foe? Critical reflections on the relevance and role of social capital in health promotion and community development. Soc Sci Med 60, 2819–2832.

Walmsey, S., Purvis, J., Ninnes, C., 2006. The role of small-scale fisheries management in the poverty reduction strategies in the Western Indian Ocean region. Ocean & Coastal Management 49, 812–833.

Walters, C., 2000. Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficacy of marine protected areas: how large should protected areas be? Bulletin of Marine Science 66, 745–757.

Wattage, P., Glenn, H., Mardle, S., Van Rensburg, T., Grehan, A., Foley, N., 2011. Economic value of conserving deep-sea corals in Irish waters: A choice experiment study on marine protected areas. Fisheries Research 107, 59–67.

Wattage, P., Mardle, S., Pascoe, S., 2005. Evaluation of the importance of fisheries management objectives using choice-experiments. Ecological Economics 55, 85–95.

Wätzold, F., Drechsler, M., 2005. Spatially uniform versus spatially heterogeneous compensation payments for biodiversity-enhancing land-use measures. Environmental and Resource Economics 31, 73–93.

Weaver, J., 2011. Carbon-rich mangroves ripe for conservation. Nature. doi:10.1038/news.2011.205

Weeratunge, N., Snyder, K.A., Sze, C.P., 2010. Gleaner, fisher, trader, processor: understanding gendered employment in fisheries and aquaculture. Fish and Fisheries 11, 405–420.

Weible, C.M., 2008. Expert-Based Information and Policy Subsystems: A Review and Synthesis. Policy Studies Journal 36, 615–635.

Wendland, K.J., Honzák, M., Portela,R., Vitale, B., Rubinoff, S. & Randrianarisoa, J., 2010. Targeting and implementing payments for ecosystem services: Opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. *Ecological Economics*, 69(11), pp. 2093-2107.

Westermann, O., Ashby, J., Pretty, J., 2005. Gender and social capital: The importance of gender differences for the maturity and effectiveness of natural resource management groups. World Development 33, 1783–1799.

White, C., Halpern, Benjamin S. & Kappel, C.V., 2012. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. Proceedings of the National Academy of Sciences.

Available at: http://www.pnas.org/content/early/2012/02/27/1114215109 [Accessed August 31, 2012].

Whittington, D., Pagiola, S., 2012. Using contingent valuation in the design of payments for environmental services mechanisms: a review and assessment. The World Bank Research Observer 27, 261–287.

Williamson, D.H., Russ, G.R., Ayling, A.M., 2004. No-take marine reserves increase abundance and biomass of reef fish on inshore fringing reefs of the Great Barrier Reef. Environmental conservation 31, 149–159.

Wilson, G.A., 1996. Farmer environmental attitudes and ESA participation. Geoforum 27, 115–131.

Woldenhanna, T., Oskam, A., 2001. Income diversification and entry barriers: evidence from the Tigray region of northern Ethiopia. Food Policy 26, 351–365.

Woolcock, 2001. The place of social capital in understanding social and economic outcomes. Canadian Journal of Policy Research 2, 11–17.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., Zeller, D., 2009. Rebuilding Global Fisheries. Science 325, 578–585.

Wossink, G.A.A., van Wenum, J.H., 2003. Biodiversity conservation by farmers: analysis of actual and contingent participation. European Review of Agricultural Economics 30, 461 –485.

Wunder, S., 2008. Payments for environmental services and the poor: concepts and preliminary evidence. Environment and Development Economics 13, 279–297.

Wunder, S., Albán, 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. Ecological Economics 65, 685–698.

Wunder, S., 2005. Payments for environmental services: Some nuts and bolts. CIFOR: Center for International Forestry Research.

Wunder, S., 2006. Are direct payments for environmental services spelling doom for sustainable forest management in the tropics? Ecology and Society 11, 23.

Wunder, S., 2007. The efficiency of payments for environmental services in tropical conservation. Conservation Biology 21, 48–58.

Wunder, S., Engel, S., Pagiola, S., 2008. Taking stock: a comparative analysis of payments for environmental services program in developed and developing countries. Ecological Economics 65, 834–852.

Wünscher, T., Engel, S., Wunder, S., 2008. Spatial targeting of payments for environmental services: a tool for boosting conservation benefits. Ecological Economics 65, 822–833.

WWF, 2004. The Eastern African Marine Ecoregion Vision: A large-scale conservation approach to the management of biodiversity. WWF: Dar er Salaam, Tanzania.

Xu, Z., Bennett, M.T., Tao, R., Xu, J., 2004. China's Sloping Land Conversion Programme four years on: current situation, pending issues. International Forestry Review 6, 317–326.

Yahya, S.A.S., 2010. *Rapid surveys of the coral reefs of northern Mtwara.* Report for CARE East African Regional Office, 31pp.

Yang, W., Liu, W., Viña, A., Luo, J., He, G., Ouyang, Z., Zhang, H., Liu, J., 2013a. Performance and prospects of payments for ecosystem services programs: Evidence from China. Journal of environmental management 127, 86–95.

Yang, W., Liu, W., Viña, A., Tuanmu, M.-N., He, G., Dietz, T., Liu, J., 2013b. Nonlinear effects of group size on collective action and resource outcomes. PNAS 201301733.

Zaman, H., 2004. The scaling-up of microfinance in Bangladesh: determinants, impact, and lessons. World Bank Publications.

Zanetell, B.A., Knuth, B.A., 2004. Participation Rhetoric or Community-Based Management Reality? Influences on Willingness to Participate in a Venezuelan Freshwater Fishery. World Development 32, 793–807.

Zbinden, S., Lee, D.R., 2005. Paying for environmental services: an analysis of participation in Costa Rica's PSA program. World Development 33, 255–272.

Zilberman, D., Lipper, L., McCarthy, N., 2008. When could payments for environmental services benefit the poor? Environment and Development Economics 13, 255–278.

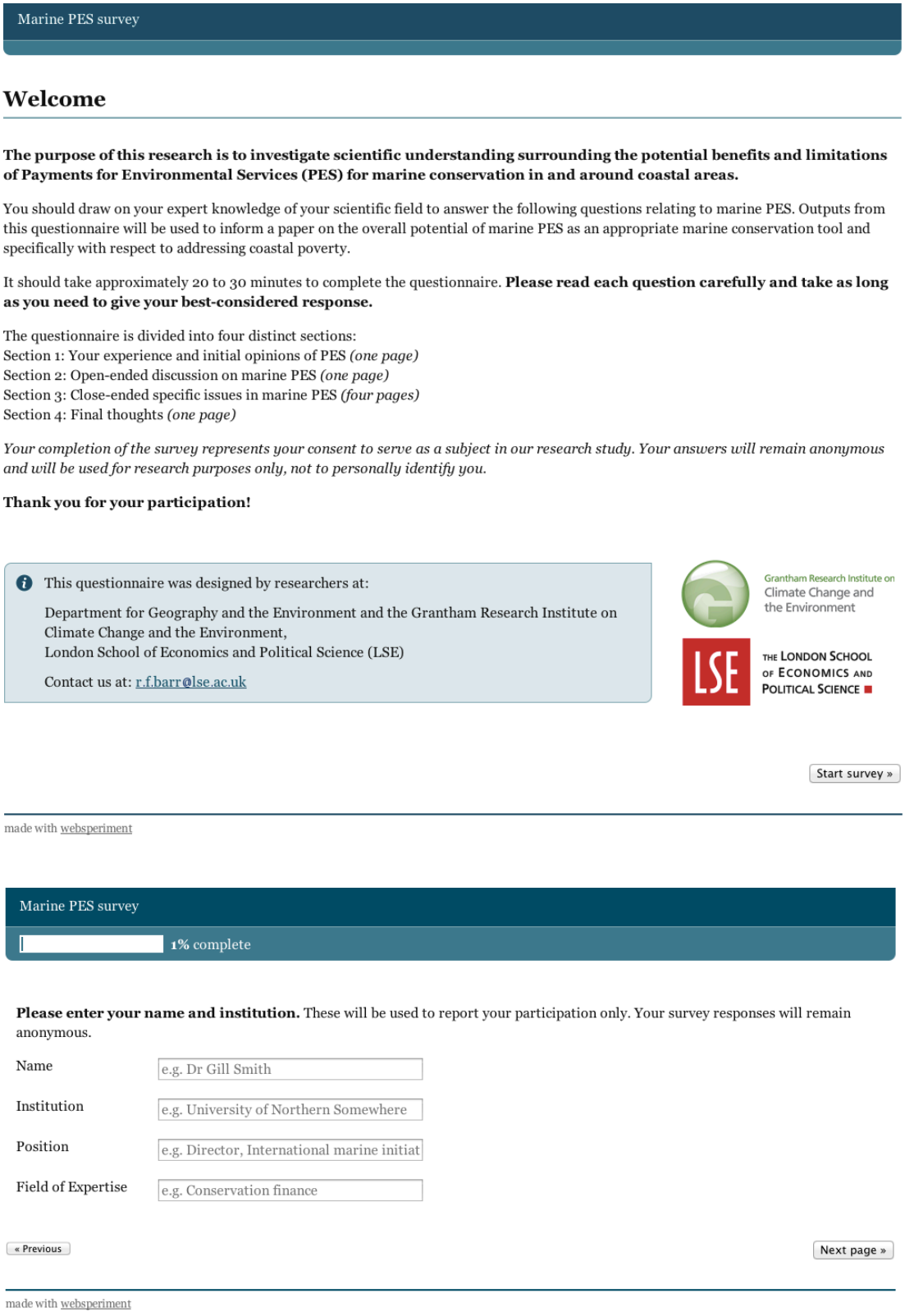
Zwick, S., Kett, H., 2010. Conservationists endorse plan to fold mangroves into REDD+ talks [WWW Document]. Eocsystem Marketplace: Forest Carbon Portal. URL http://www.forestcarbonportal.com/content/conservationists-endorse-plan-fold-mangroves-redd-talks

Appendix A

Supplementary information for expert elicitation

A1 Web survey presented to marine and PES experts

The following screenshots display all pages of web survey except where links are hidden (in blue).



Marine PES survey

Welcome

The purpose of this research is to investigate scientific understanding surrounding the potential benefits and limitations of Payments for Environmental Services (PES) for marine conservation in and around coastal areas.

You should draw on your expert knowledge of your scientific field to answer the following questions relating to marine PES. Outputs from this questionnaire will be used to inform a paper on the overall potential of marine PES as an appropriate marine conservation tool and specifically with respect to addressing coastal poverty.

It should take approximately 20 to 30 minutes to complete the questionnaire. **Please read each question carefully and take as long as you need to give your best-considered response.**

The questionnaire is divided into four distinct sections:
Section 1: Your experience and initial opinions of PES *(one page)*
Section 2: Open-ended discussion on marine PES *(one page)*
Section 3: Close-ended specific issues in marine PES *(four pages)*
Section 4: Final thoughts *(one page)*

Your completion of the survey represents your consent to serve as a subject in our research study. Your answers will remain anonymous and will be used for research purposes only, not to personally identify you.

Thank you for your participation!

This questionnaire was designed by researchers at:

Department for Geography and the Environment and the Grantham Research Institute on Climate Change and the Environment,
London School of Economics and Political Science (LSE)

Contact us at: r.f.barr@lse.ac.uk

Grantham Research Institute on Climate Change and the Environment

THE LONDON SCHOOL OF ECONOMICS AND POLITICAL SCIENCE

Start survey »

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Marine PES survey

1% complete

Please enter your name and institution. These will be used to report your participation only. Your survey responses will remain anonymous.

Name e.g. Dr Gill Smith

Institution e.g. University of Northern Somewhere

Position e.g. Director, International marine initiat

Field of Expertise e.g. Conservation finance

« Previous

Next page »

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Marine PES survey

3% complete

Section 1: Experience and opinions of PES

1. Have you worked on implementing a resource conservation program on the ground? If so how many? *(This may include involvement in the design phase as well as application)*

- No
- 1-4
- 5-9
- 10 or above

2. Have you worked on implementing a ***marine or coastal*** resource conservation program on the ground? If so how many? *(This may include involvement in the design phase as well as application)*

- No
- 1-4
- 5-9
- 10 or above

3. Overall, how closely do you follow the academic research developments within the PES literature? *(e.g. peer-reviewed journal articles, book chapters)*

- Do not follow
- Follow very occasionally
- Somewhat follow
- Closely follow
- Very closely follow

4. How many peer-reviewed papers have you authored or co-authored about or involving PES? *(e.g. journal articles, book chapters)*

- None
- 1-4
- 5-9
- 10 or above

5. Overall, how familiar would you say you are with applied PES instruments? *(e.g how well do you understand the issues relating to PES implementation in the field)*

- Not at all familiar
- A little familiarity
- Average familiarity
- Good familiarity
- Very good familiarity

6. How many PES programs have you worked on implementing on the ground? *(Again, this may include involvement in the design phase as well as application)*

- None
- 1-4
- 5-9
- 10 or above

7. Please tick the criteria that you believe a PES instrument must meet in order to be considered a true-PES scheme:

*Please tick **all** that apply*

- ☐ the PES scheme must involve a well-defined environmental service
- ☐ the environmental service to be purchased must be purchased by at least one service buyer
- ☐ the environmental service to be provided must be provided by at least one service provider
- ☐ the parties involved in the PES transaction must be involved in a voluntary capacity
- ☐ the PES payment must be conditional on environmental service provision
- ☐ the incentive offered for the environmental service must be positive
- ☐ the incentive offered for the environmental service must be cash
- ☐ the scheme must provide environmental services to a level above those already provided without the program (i.e. must be additional)
- ☐ the environmental service must be provided by service providers with well established property rights
- ☐ the PES scheme must be adequately enforced
- ☐ none of the above

---

8. Do you consider there to be other criteria for a true-PES scheme not covered above?
If so please list below:

« Previous Next page »

Marine PES survey

4% complete

Marine Payments for Environmental Services

We are about to ask you about your beliefs regarding marine PES. PES mean many different things to different people. In order to gain consistent results we define PES broadly as per Sommerville 2009

PES are defined as approaches that (1) transfer positive incentives to environmental service providers that are (2) conditional on the provision of the service.

Marine PES programs can include the following environmental services:

- shoreline stabilisation
- marine and coastal carbon storage and sequestration
- fish nursery habitats
- marine species, habitat and biodiversity conservation
- coastal water quality and pollution filtration

As an example, we consider a marine PES scheme to be one whereby fishermen are paid to avoid fishing within a zoned area (e.g. a fish-nursery site) and payments are made on a monthly basis conditional on the action of not fishing within this zone. Buyers for the service may include industry, governmental or non-governmental organisations.

We do not consider to be a marine PES scheme a scenario similar to the one above but with the exception that fishers receive only a one-off payment. This is because all conditionality of the payment for ecosystem service provision is removed. In the same vein, schemes such as boat and licence buyback schemes are not considered to be PES schemes.

What are buyback schemes »

When answering the subsequent questions relating to marine PES we would like you to think of a marine PES instrument as defined in the previous paragraphs.

▸ Were you previously familiar with the information presented on this page?

- ○ Not at all familiar
- ○ Somewhat familiar
- ○ Familiar
- ○ Very familiar

« Previous

Next page »

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Marine PES survey

12% complete

Section 2: Marine PES

We would first like to ask your views about three distinct aspects of marine PES programs. These are open-ended questions that will be coded for analysis. The results we can obtain will very much depend on the level of information you provide. Please take the time to think about your answers and elaborate on these; we very much want to hear all your beliefs and concerns on these matters. Please feel free to take as much time as you wish. We anticipate that to fully engage with these questions should take approximately 10 to 15 minutes.

1. Over the last few years a great deal of lessons have been learnt in terrestrial PES schemes with respect to improving delivery, efficiency and implementation.

1.a. What do you believe are the benefits of bringing PES to the marine environment? *Where possible please support your arguments by applying these to your own experience.*

I think...

1.b. What do you believe are the challenges of bringing PES to the marine environment? *Again where possible please support your arguments by applying these to your own experience.*

I think...

2. Payments for Environmental Services are a fairly new instrument in the conservation portfolio, marine PES to an even greater degree.

2.a. Do you believe marine PES have the potential to better protect the marine and coastal environment over other marine conservation tools currently in use such as community-based management programs, marine protected areas and gear exchange/buy-back schemes? *(e.g. what are the advantages and limitations of marine PES over these other more established instruments?)*

I think...

2.b. Do you believe marine PES have the potential to be complementary instruments within the conservation portfolio?

I think...

3. Coastal areas often experience high levels of poverty, have complex poverty dynamics, and show high levels of reliance by the poor on coastal and marine resources.

PES schemes can be designed to address issues of poverty. One of the main benefits of PES schemes is considered to be their efficiency and cost-effectiveness over other less direct conservation instruments. PES efficiency will likely be reduced under a pro-poor PES design (i.e. due to higher transaction costs, economies of scale, etc).

PROPOOR PES: a PES program which maximises its potential positive impacts and minimises its potential negative impacts on the poor.

3.a. To what extent should a marine PES scheme explicitly attempt to address poverty and be pro-poor by design? Please provide reasoning with your answer

I think...

3.b. Do you think marine PES schemes are not explicitly pro-poor will effect poverty levels, both absolute and relative, in these coastal areas?

I think...

« Previous

Next page »

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Marine PES survey

63% complete

Section 3: Marine PES within the marine conservation portfolio

We would now like follow up the open-ended questions by asking your opinion regarding some specific attributes that the scientific literature has identified as possible advantages and disadvantages of PES schemes over other marine conservation instruments, such as community-based management, marine protected areas and gear exchange/buy-back schemes.

1. Please consider the following statements carefully. For each, please indicate whether you strongly disagree, disagree, neither agree nor disagree, agree, or strongly agree with respect to marine PES schemes ***over other marine conservation instruments.***

A box has been provided at the bottom of this page for any follow up explanations or comments you wish to enter. Please be sure to specify to which question comments relate.

1.a. Marine PES that are financed by demand from environmental service buyers will result in better targeting of priority areas *compared to other marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.b. Marine PES that use cash to incentivise behavioural change will improve sustainable use of marine resources by local resource users *compared to other marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.c. Payment within a marine PES scheme that is conditional on environmental service delivery will better guarantee delivery of environmental goods and services *compared to other marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.d. Marine PES create direct markets between environmental service providers and sellers and so promote more cost-effective conservation *compared to other marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.e. Marine PES payments made for services above and beyond those already provided will increase environmental service provision (i.e.additionality) *compared to other marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.f. Marine PES will require more well defined seascape property rights to adequately protect marine resources within community landscapes *compared to other marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.g. Incentives/payments made through marine PES schemes have a greater potential to promote income diversification over other non-incentive based *marine conservation instruments.*

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.h. Marine PES payments are repeated and not one-off and so will guarantee long-term conservation of the environmental services in question *compared to other marine conservation instruments*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.i. The multiple and fragmented nature of environmental service providers (such as fishing communities) will make these PES schemes harder to implement *compared to other marine conservation instruments*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.j. Measurement and accountability of marine environmental services will make these PES schemes more costly to implement *compared to other marine conservation instruments*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.l. Marine PES can create more perverse incentives *compared to other marine conservation instruments*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.m. other? If you wish to add something which you feel we have missed please do so here

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

1.n. other? If you wish to add something which you feel we have missed please do so here

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

If you wish to add an explanation to any of your answers above please do so here.

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« Previous

Next page »

Marine PES survey

71% complete

Section 3: Marine PES within the PES portfolio

We would now like to ask you a few questions relating to marine PES schemes in relation to terrestrial PES schemes.

Problems identified as important to terrestrial PES schemes may, or may not, relate to marine PES schemes. Please now consider which characteristics you believe would be more problematic within a marine PES compared with terrestrial PES schemes, (e.g. hydrological PES programs where by upstream water-users pay individulas downstream to improve soil management or carbon PES programs where by global beneficiaries pay local forestry users to reduce deforestation)

2. Please consider the following statements carefully. For each, please indicate whether you strongly disagree, disagree, neither agree nor disagree, agree, or strongly agree with respect to marine PES schemes ***over terrestrial PES schemes.***

A box has been provided at the bottom of this page for any follow up explanations or comments you wish to enter. Please be sure to specify to which question comments relate.

2.a. The more complex nature of marine environments will result in lower demand for marine PES by service buyers *compared with terrestrial PES schemes*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.b. The use of cash as an incentive will pose additional problems for communities within a marine PES scheme *compared to those within terrestrial PES schemes*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.c. Monitoring and guaranteeing additionality (i.e. payments are made for services above and beyond those already provided) will prove more difficult for marine PES *compared with terrestrial PES schemes*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.d. Defining property rights within the seascape in order to adequately protect marine resources will prove more challenging within the marine context *compared with a terrestrial PES context*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.e. Incentives/payments made through marine PES schemes have a greater potential to promote income diversification within coastal communities *compared with those within terrestrial PES schemes*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.f. Enforcing PES requirements will prove more difficult within a marine context *compared with a terrestrial PES context*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.g. Given the complex poverty dynamics within coastal communities marine PES have a higher likelihood to increase relative poverty for some members within these communities *compared with a terrestrial PES context*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.h. Incentives/payments made through marine PES schemes have a greater potential to promote perverse incentives within coastal communities *compared to those within terrestrial PES schemes*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.i. The multiple and fragmented nature of environmental service providers (such as fishing communities) will make marine PES more difficult to implement *compared with a terrestrial context*.

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.j. other? If you wish to add something which you feel we have missed please do so here

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

2.k. other? If you wish to add something which you feel we have missed please do so here

Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
○	○	○	○	○

If you wish to add an explanation to any of your answers above please do so here.

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« Previous Next page »

Marine PES survey

78% complete

Section 3: PES and risks to the poor

Given the high prevalence of poverty in coastal areas we are particularly interested in your views on how suitable and effective marine PES schemes could be in addressing the concurrent issues of environmental protection and poverty alleviation.

The literature has identified the following as potential problems for ***poor, marginalised and vulnerable groups*** within PES schemes. Do you consider these to be problems within a marine PES scheme? And if so what level of risk do these issues pose for those poor, marginalised and vulnerable groups?

A box has been provided at the bottom of this page for any follow up explanations or comments you wish to enter. Please be sure to specify to which question comments relate.

PROPOOR PES: a PES program which maximises its potential positive impacts and minimises its potential negative impacts on the poor.

Please note: poverty does not equal equity. Poverty refers to a pronounced deprivation of wellbeing and is a multi-dimensional concept relating to the economic, social and political exclusion of certain members of society. Equity, on the other hand, relates to the distribution of socio-economic factors and goods within society according to an agreed set of principles. These can, in turn, influence the allocation of PES outcomes or an individual's ability to participate in a PES scheme.

3. For the issues mentioned below please consider the implications at the 'on-the-ground' community level.

3.a. Marine PES may lead to an increase in relative poverty between participants within the scheme.

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.b. Marine PES may lead to an increase in relative poverty between participants within the marine PES scheme and those non-participants.

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.c. Marine PES can lead to a loss of defacto property rights of marginalised groups.

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.d. The injection of incentives/payments into coastal communities through marine PES schemes may lead to a degradation of existing social capital ties.

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.e. Marine PES schemes may increase relative poverty through increasing the price of certain commodities.

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.f. Marine PES can lead to monopolisation of environmental services by elite groups.

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.g. other? If you wish to add something which you feel we have missed please do so here

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

3.h. other? If you wish to add something which you feel we have missed please do so here

○ No risk ○ Low risk ○ Moderate risk ○ High risk ○ Extremely high risk

If you wish to add an explanation to any of your answers above please do so here.

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« Previous

Next page »

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Marine PES survey

85% complete

Section 3: High risk groups

Some socioeconomic groups may be more at risk from marginalisation within a marine PES scheme than others.

4. Now consider the following socioeconomic groups and their level of risk of marginalisation from a marine PES scheme. Please indicate what you believe to be their level of risk of marginalisation.

Again, a box has been provided at the bottom of this page for any follow up explanations or comments you wish to enter. Please be sure to specify to which question comments relate.

4.a. **women** vs. men

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.b. **dedicated fishers** *(fishing is main, often only livelihood)* vs. fishers with a wider portfolio of activities (e.g. fisher-farmers)

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.c. **fisher-farmers** *(fishing is part of a wider livelihood portfolio but remains a very important activity and for some the only/primary source of cash)* vs. non-fishers

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.d. **migrant fishers** *(dedicated non-local fishers who set up semi permanent camps and follow fish catch based on seasonality)* vs. local fishers

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.e. **illegal fishers** *(fishers who fish using illegal fishing gear either to increase profits or due to inability to adopt legal gears)* vs. legal fishers

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.f. **poor non-fishing farmers** *(farmers who lack the means to fish)* vs. those with the means to fish

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.g. **upstream vendors of marine products** vs. those vendors whose businesses are not reliant on marine products

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.h. **elderly** vs. young adults

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.i. **children** vs. young adults

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.j. **those poor with high consumptive dependence on fish products** vs. those with lower dependence (e.g. can choose from a wider choice of alternatives)

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.k. fishers from **female-headed households** vs. male-headed households

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.l. other? If you wish to add something which you feel we have missed please do so here

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

4.m. other? If you wish to add something which you feel we have missed please do so here

Much less at risk of marginalisation	Less at risk of marginalisation	No more or less at risk of marginalisation	More at risk of marginalisation	Much more at risk of marginalisation
○	○	○	○	○

If you wish to add an explanation to any of your answers above please do so here.

« Previous

Next page »

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Marine PES survey

93% complete

Section 4: Marine PES overview

Thank you for your time. In light of the past questions raised and you responses we have **three** final questions we would like you to take a look at:

1. On balance, do you agree with the statement: I believe marine PES to be a suitable instrument for marine conservation

- ○ No, I strongly disagree
- ○ No, I somewhat disagree
- ○ I neither agree nor disagree
- ○ Yes, I somewhat agree
- ○ Yes, I strongly agree

If you wish to add an explanation to your answer please do so below

I think...

2. Are there conditions that you see as being absolutely essential for a marine PES to function successfully?

I think...

3. Can you give an example of where and under what conditions you see marine PES instruments being most appropriate for marine conservation? And, if relevant, do you believe a marine PES scheme has the potential to function on the ground in those areas in which you have worked? *If so, please identify the area and circumstances in which you work/worked, and why you feel it has the potential to work or not.*

I think...

(Optional) Please enter your e-mail address here if you would like to be informed about the results of this survey and/or be willing to answer any follow-up questions in the future. This information will be kept confidential.

E-mail address e.g. G.Smith@university.edu

« Previous Next page »

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Marine PES survey

100% complete

Thank you

THE END

Thank you for taking the time to complete this survey.

If you have any further comments about this questionnaire, please contact Rhona Barr at r.f.barr@lse.ac.uk.

Thank you for your participation!

A2 List of participating experts and organisations

Id.	*Name*	*Organisation*
1	Andrew Bovarnick	United Nations Development Program
2	Angelica Klaussén	Stockholm University, Sweden
3	Beatriz Lucas	Comunidad y Biodiversidad, Mexico
4	Thomas Binet	CEMARE, University of Portsmouth
5	Charlene Watson	Overseas Development Institute
6	Charles Palmer	London School of Economics
7	Chris Williams	New Economics Foundation (NEF)
8	Diane Davidson	Legal Ray Consultants, LLC
9	Emily Gaskin	National Oceanic and Atmospheric Administration (NOAA)
10	Essam Yassin Mohammed	International Institute for Environment and Development (IIED)
11	James Spurgeon	Sustain Value
12	Jason Krumholz	University of Rhode Island
13	Jay Udelhoven	The Nature Conservancy (TNC)
14	Jeff Kinch	National Fisheries Authority
15	Jos Hill	Environmental Defense Fund
16	Josh Donlan	Advanced Conservation Strategies
17	Joshua Cinner	James Cook University, Australia
18	Justin Bousquin	EPA ORD Contractor
19	Katherine Short	ICL/WWF
20	Keith Lawrence	Conservation International
21	Kristina Raab	Wageningen University, Netherlands
22	Lad Atkins	REEF Environmental Education Foundation
23	Linwood Pendleton	Duke University
24	Mark Ellis-Jones	Ecosystem Equity
25	Matt Sommerville	Tetra Tech ARD
26	Matthew Cranford	London School of Economics
27	Melita Samoilys	CORDIO East Africa
28	Nick Hill	Zoological Society of London
29	Patricia Bradley	U.S. Environmental Protection Agency
30	Patrick Christie	University of Washington
31	Paul Collins	Coral Reef Research Foundation
32	Paul Ferraro	Georgia State University
33	Peter May	University of Washington
34	Phanor H Montoya-Maya	University of KwaZulu-Natal, South Africa
35	Pilar Herran	Independent
36	Rashid Sumaila	University of British Columbia
37	Sian Morse-Jones	Fauna & Flora International (FFI)
38	Stefano Pagiola	World Bank
39	Sven Wunder	CIFOR
40	Tim McClanahan	Wildlife Conservation Society (WCS), Kenya
41	Todd Gartner	World Resource Institute (WRI)
42	Tom Clements	Wildlife Conservation Society (WCS), UK

Appendix B

Supplementary information for case study

B1 Data sheets for the questionnaires presented to fishers

Section A. Tracking Information

A1. Date form filled (dd/ mm/ yyyy) | | | | | | | | |

A2. Name of the enumerator _____

A3. Form filled at (Village/Location)
A4. How long have you lived in this village? years

Section B. Socio-demographic Information

B1. Name of respondent:	**B2**. Age:
B3. Married?:	
B4. Member of fishing group?	
If Yes name:	

B5. Respondent's gender	☐ **1** Male	☐ **2** Female
B6. Respondent's highest level of education	☐ **1** None ☐ **2** Primary ☐ **3** Secondary ☐ **4** High school	☐ **5** College / Univ. ☐ **6** Technical ☐ **7** Other (specify) _____
B7. Are you the head of the household?	☐ **1** Yes	☐ **2** No
B8. If responded no to question B7. Is the head of your household	☐ **1** Male ☐ **2** Married female	☐ **3** Female spinster ☐ **4** Widow female
B9. Head of household's highest level of education	☐ **1** none ☐ **2** primary ☐ **3** secondary ☐ **4** high school	☐ **5** college / univ. ☐ **6** technical ☐ **99** other (specify) _____
B10. How many in your household? (share food and income) **Fill in number in corresponding boxes**	☐ **1** Adult males ☐ **2** Adult females	☐ **3** Male children ☐ **4** Female children
B11. Is your <u>main</u> occupation fishing?	☐ **1** Yes	☐ **2** No
B12. How long have you been a fisherman? Tick one.	☐ **1** Less than 1 year ☐ **2** 1-2 years ☐ **3** 3-5 years ☐ **4** 6-10 years ☐ **5** 11-15 years ☐ **6** 16+ years	

B13. Given that your income from fishing changes on a daily basis but thinking about good and bad days how much would you say you earn from fishing on:

a) on a **good** day? _____
b) on a **average** day? _____
c) on a **bad** day? _____

d) this income is split between how many? _____ and is for what kind of fishing? _____

What other types of fishing do you do?

B14. On average how many days a week do you fish? Time in hours?

b. Would you say you experience more good or bad days or the same?

Section C. Aspirations for the future and attitudes

C1. Why did you become a fisher?

C2. Do you enjoy being a fisher?

☐ 1 Not at all ☐ **2** I don't like it very much ☐ **3** I neither like nor dislike it ☐ **4** I like it ☐ **5** I like it very much

Please explain why:

C3a. Are there other activities available to you that you would consider doing? ☐ **1** Yes ☐ **2** No

b. If yes please list the professions:

C4. If you had the chance, would you prefer to do one of these activities as your main activity? ☐ **1** Yes ☐ **2** No

b. Please explain why:

C5. Do you think you will still be fishing in 5 years time? ☐ **1** Yes ☐ **2** No

b. Please explain why:

C6. Overall, do you think the number of fish you catch on the reef has changed over the last five years? ***N.B. Enumerators: Please clarify this is yearly trends not seasonal***

☐ **1** Decreased a lot ☐ **2** Decreased a little ☐ **3** It is about the same ☐ **4** Increased a little ☐ **5** Increased a lot

b. Please explain why:

C7. Overall, how do you think your income from fishing has changed in the last five years?

☐ **1** Decreased a lot ☐ **2** Decreased a little ☐ **3** It is about the same ☐ **4** Increased a little ☐ **5** Increased a lot

b. Please explain why:

C8. Overall, do you think the number of days you are required to fish to catch the same amount has increased or decreased since you started working as a fisherman?

☐ **1** These days I spend fewer days ☐ **3** The same ☐ **5** The days I spend more days

b. Please explain why:

C9. Overall, do you think there are more fishermen now than there were five years ago?

☐ **1** There are many more ☐ **2** There are a few more ☐ **3** It is about the same ☐ **4** There are a few less ☐ **5** There are far fewer

b. Please explain why:

C10. Overall, do you think you are better off, worse off or about the same than before?

☐ **1** A lot worse off ☐ **2** A little worse off ☐ **3** The same ☐ **4** A little better off ☐ **5** A lot better off

b. Please explain why:

C11. Overall, do you think the marine park can be beneficial to you in the future?

☐ **1** Not at all ☐ **2** Not very much ☐ **3** I don't know ☐ **4** A little ☐ **5** A lot

b. Please explain why:

C12. Would you be happy if your son(daughter) became a fisherman(tandilo fisher)?

☐ **1** Very unhappy ☐ **2** Unhappy ☐ **3** I don't know ☐ **4** Happy ☐ **5** Very happy

b. Please explain why:

C13. Do you agree with the statement: There will always be enough fish in the sea available for people to fish?

☐ **1** Totally disagree ☐ **2** Disagree a little ☐ **3** Neither disagree or agree ☐ **4** Agree a little ☐ **5** Totally agree

b. Please explain why:

C14. Do you agree with the statement: If the reef disappears if will make no change to my life?

☐ **1** Totally disagree ☐ **2** Disagree a little ☐ **3** Neither disagree or agree ☐ **4** Agree a little ☐ **5** Totally agree

b. Please explain why:

C15. In your opinion, what kind of improvements could be made to the park regulations?

D. VALUATION

D1. ***[Enumerator please read this passage to the interviewee]***

Due to growing numbers of people both locally and globally, there is an increasing dependence on fish. Increased populations and demand has led to an increase in the number of people fishing. This has resulted in a decline of fish in the area. Fishing also continues in areas of high importance for fish breeding. Fishing of these nursery sites will lead to fish stocks further declining.

There are a number of breeding grounds within the marine park. These areas are important for the restocking of fish populations and maintaining fish numbers within the park. Furthermore, by protecting these areas which are important to juvenile and breeding fish, fish stocks may

recover and increase from their current levels, for this reason WWF and CARE International are interested in protecting these areas while at the same time improving the livelihoods of local fishermen

Enumerator show fisherman map with marked off areas and explain that these areas would be areas in which fishing would not be permitted Also make sure without doubt that the respondent understands where these areas are. Ask him some land marks to confirm

Please consider how much you fish within these marked areas and the proportion of fish you catch there.

D1a. How much of your total fishing time do you spend fishing within these zoned areas?

☐ **1** None ☐ **2** Less than half ☐ **3** About half ☐ **4** More than half ☐ **5** All

b. How many days would you say you fish in these areas each week/month?

D1b. How much of your catch comes from these areas?

☐ **1** None ☐ **2** Less than half ☐ **3** About half ☐ **4** More than half ☐ **5** All

D1c. If the regulations changed and the law prevented you from fishing in these areas would you change your fishing behaviour?

☐ **1** Yes
☐ **2** No
☐ **3** Don't know

b. Please explain why:

D1e. If the regulations changed and the law prevented you from fishing in these areas would this have implications for you welfare?

☐ **1** Yes
☐ **2** No
☐ **3** Don't know

b. Please explain why:

D1f. How would your welfare change exactly?

☐ 1 Welfare would decrease
☐ 2 Welfare would stay the same
☐ 3 Welfare would increase

b. Please explain why:

E2. Enumnerators please read the following to the respondent explaining the map again:

CARE International, alongside WWF, as previously mentioned are interested in improving the marine environment in the area, as well as supporting the local livelihoods, particularly of the fishers who rely on these resources. In order to do so CARE International are considering a conditional cash transfer programme. A conditional cash transfer programme would mean that community fishers would be asked to *not fish in those specific zoned areas* as shown in the map as well as *stop the use of all illegal fishing gear in all areas*, but also compensated for their loss of earnings from these changes.

Enumerators: Again show the respondent the map, identifying all the areas where fishing would not be permitted, also explain the scenario as described below.

The scheme would originally run for 4 to 5 years. The payments would be conditional upon *all fishers within this community, NOT fishing in these designated areas or using illegal gear*. The payments would be made on a monthly basis, and all payments would be *cancelled* if fishing continued within the designated areas and illegal gears continued to be used. During this time additional investment will be made into the development of alternative occupations which will increase the availabilities of alternative activities in the area. Monitoring would be a combined effort between local communities, who would all lose out if the rules were broken and the Marine Park authority inside park/BMUs officials outside park. In this time it is expected that fish stocks will have suitable recovered and fishing profits increased, as well as management practices improved allowing the long-term sustainability and profitability of the fishing.

For example far away in the Pacific Ocean, closing areas to fishing has increased local fishers catch, both in size and amount. The improvements took a few years to be seen, as fish within these areas require time to mature and grow, however after this they often leave the protected area. This is why CARE International is looking to support the programme with conditional payments over the time it is required for the stocks to recover. However, now these small fishing communities in the Pacific which have seen larger catches, more fish and higher income than neighbouring communities without such protected areas.

E1a. Do you think a conditional cash transfer programme – *where you are paid a certain amount of money each month for a number of years, but at the same time must stop fishing within the zoned areas and all illegal fishing during this time period* – is something you would consider enrolling in?	☐ **1** Yes ☐ **2** No ☐ **3** Don't know ☐ **4** Depends on the amount ☐ **5** Depends on what others would do
b. Can you tell me the reason for your response:	

If E2. answered No – go to Section F, If YES – continue with Section E.

E3. Now I would like you to consider how much is your income as well as the costs associated with fishing in these areas. You mentioned that you earned _____(insert answer from B12) from fishing. You also mentioned that you caught _____ (insert answer from E1b) of the fish in these areas.

Considering that these areas only provide a certain amount of your fish catch and that this time can be spend fishing elsewhere would you accept the following weekly payment to stop fishing in these areas? I am asking for the minimum amount you would be prepared to accept, that is, the amount that would leave you as well off as you are today, not better, not worse?

NB: READ FIRST.

It's important to realize that these are early studies. To help us determine if this plan will help and is possible, it is important that you give us truthful facts, i.e. the true minimum level of compensation that you would need to cover your opportunity costs only for the loss of fishing in these areas.

Please consider how much you catch in these areas and the time you spend there compared to catch and time in other areas. Please note you can continue to fish in other areas using legal gear and generate income from these areas. The feasibility of this programme going ahead will depend on if it is affordable, it is possible that overinflated responses can lead to the failure of this initiative.

NOW ASK

Considering all of the above and your fishing patterns what is the minimum weekly compensation value you would accept to stop fishing in these areas.

Enumerators go down list asking if they would accept each value in sequence. Stop when the responded says yes, and writing yes in appropriate box

Sh T	Yes/No
500	
700	
1000	
1500	
2000	
3000	
5000	
7500	
10,000	
12,500	
15,000	
20,000	
30,000	
Other	

E4a. Again, you said that you fish _____ days per week/month in these areas (D1a). By using your averge income (B12) I calculated that gives you _____ TSh per week / month. This value appears to be more/less than what you asked for in compensation.

b. Why is this?

E5a. Suppose other fishermen asked for roughly the same level of compensation as you, but the scheme was unable to cover all these costs, would you be willing to lower the level of compensation required?

☐ **1** Yes
☐ **2** No
☐ **3** Don't know
☐ **4** Depends on the amount

b. What new level of compensation would you require? _____

TSh

c. Please explain why:

E5a. From our research and talking to other fishers we see that the income needed to compensate fishers for loss of fishing in these areas is 1000TSh a day, or 5000TSh a week. Understanding this would you now be willing to lower the level of compensation required?

☐ **1** Yes
☐ **2** No
☐ **3** Don't know

b. What new level of compensation would you require? _____

TSh

c. Please explain why:

F. CHOICE EXPERIMENT – FISHERMEN ONLY

F1. Now we would like to ask you a little more about the types of restrictions you would prefer to see within the your fishing area.

I want you to think about the current law and about further prohibitions in your fishing area, more specifically the introduction of additional no-take zones and the prohibition of certain gears. These changes come with compensation for these additional restrictions.

I am going to show you three choice cards. Two cards will show you new fishing regulations and the third card shows you the current regulation in your fishing area.

Each card has two attributes relating to the possible changes in law which can change:

- The percentage of your current fishing area to be closed to fishing
- The allowable net mesh size (in inches)

The final attribute on these cards is a monetary value. This is the level of compensation per week you would receive if these restrictions were put in place. Please remember, the values shown in BLUE are payments you would receive. Values shown in RED are payments you would make each week to have the new restrictions put in place.

Monitoring and enforcement would be a collaboration between the community and the Marine Park Authority/BMU. Payments would be made monthly and all payments would be withdrawn if the restrictions were not followed.

Please consider carefully which of the scenarios on the cards you prefer, thinking about how each restriction would effect your fishing catch, the compensation you would receive and the trade-offs between the three.

Enumerators: Please show an example of the cards and choice set and take them through the trade offs required: see below.

For example, would you prefer restrictions shown on card 1 which allowed you to fish in all areas but you were only allowed to you 6" meshed nets in these areas and were paid 10,000 TSh compensation a week or would you prefer card 2 which closes 25% of your current fishing grounds but allows the use 3" nets and rewards 5,000 TSh a week? Or do you prefer to stick with the current management restrictions as shown in card 3?

Enumerators: If interviewee understands the concept continue with game.

Enumerators: Please write card no. of cards picked from bag in table and circle choice interviewee make

Round	***First card***	***Second card***	***No change***
1.			*No change*
2.			*No change*
3.			*No change*
4.			*No change*
5.			*No change*
6.			*No change*

G. SOCIAL CAPITAL

G1. If there is a decision to be made in your community, are you involved in that decision? **b.** If Yes, how so? Actively/passively	☐ **1** Yes ☐ **2** No
G2. Are you involved in the decision about marine resource use (fishing, shell collection, etc) or management? **b.** If Yes, how so? Actively/passively?	☐ **1** Yes ☐ **2** No
G3. Are you involved in community decisions about marine resource use (e.g. fishing, shell collection, etc) or management? **b.** If Yes, how so? Actively/passively?	☐ **1** Yes ☐ **2** No

G4. How important would you say the following were in your decision to use the fishing gear you use?

	Not at all important	Not very important	Indifferent	Important	Very important	I don't know
a. skills required						
b. ease of use						
c. boat required						
d. cost of gear						
e. tradition						
f. potential returns						
g. social acceptability						
h. environmental impact						
i. management restrictions						
j. other List:						

Please list the three most important factors in choosing your fishing type:

1.

2.

3.

G5. To what extent do you trust the following groups of people?

	Not at all	Distrust more people than trust	Trust about half the people	Trust more people than distrust	Trust all	Don't know/ NA
a. people you work with						
b. people in your village						
c. community leaders						
d. local government officials						
e. Village Liasion Committee						
f. NGO staff						
g.fisheries/marine park enforcement/BMU officers						
h. fishers from other villages						
i. people using different gear						

G6. Do you belong to any groups or societies? ☐ 1 Yes ☐ 2 No

If Yes, please enter into table below:

Name of group	Group type	No. of meetings in last 6 months	No. of meetings you attended

G7. Do you have close family outside of this village? ☐ **1** Yes ☐ **2** No

If Yes, please fill in the table below for all close family members outside of this village

Relation	Live where now?	Lived in village before?	If yes, when did the person move away?	If yes, why did the person move away?

G8. With whom can you discuss important matters? *(anything important to you)*

Write the details in the table below.

Name	Occupation	Lives in village? (Y/N)	Type of information exchanged?

G9. If you noticed changes in the natural environment (e.g. the number of fish caught, the condition of the mangrove forest or reef, etc), who would you discuss this with?

Write the details in the table below.

Name	Occupation	Lives in village? (Y/N)	Type of information exchanged?

G10. Do you exchange information with anyone which is useful for you to carry out your common occupation? (E.g. you tell or are told about good fishing spots, practices, equipment, timing and season, etc?) ☐ **1** Yes ☐ **2** No

Write the details in the table below.

Name	Occupation	In village? (Y/N)	Relation	Known how long?	Meet how often?	Type of information exchanged	I give	I receive

G11. Is there any person(s) on whom you depend, or depend on you, to carry out your (their) occupation? (e.g. do you need someone else's boat gear, nets etc to carry out your occupation?) ☐ **1** Yes ☐ **2** No

Write the details in the table below.

Name	Occupation	In village? (Y/N)	Relation	Known how long?	Meet how often?	What is the exchange/ common link?	I give	I receive

G12. During bad fishing times, when catch is low, is there any person(s) on whom you can rely to help you through these weeks/months, or depend on you, to help them through difficult times? ☐ **1** Yes ☐ **2** No

Write the details in the table below.

Name	Occupation	In village? (Y/N)	Relation	Known how long?	Meet how often?	What is the exchange/ common link?	I give	I receive

G13. In your occupation do you sell your catch to anyone in particular? ☐ **1** Yes ☐ **2** No

Write the details in the table below.

Name	Occupation	In village? (Y/N)	Relation	Known how long?	Meet how often?	What is the exchange/ common link?	I give	I receive

H. FISHING DEMOGRAPHICS

H1. What gears do you and your household use?
Fill in all that apply

Gear	Tick all	Tick if respondent uses	No. of others in HH using gear	No. of gear rented	No. of gear owned with others	No. of gear owned alone
Gleaning	1					
Spear/stick	2					
Speargun	3					
Line	4					
Basket trap	5					
Tidal weir	6					
Net>=3”	7					
Net 2”-2.5”	8					
Net1.5”	9					
Net 1”	10					
Ring net (Juya)	11					
Beach seine Kokoro)	12					
Tandillo	13					
Longline	14					

H2. Do you use a boat to fish? ☐ **1** Yes ☐ **2** No

b. If Yes, what type and what level of ownership do you have?

	Tick	Boat type	How many?	Has engine?
Work on ‘Tajifi’	1			
Borrow boat	2			
Rent boat	3			
Own boat with others	4			
Own boat	5			

H3. Please describe your catch and income from catch
Gear:_____

	On a good day	On an average day	On a bad day	Split between how many?
Fish species generally caught				
Approx no/weight				
Approx income (TSh)				

H4. Did you previously use gear/nets which are now illegal? ☐ 1 Yes ☐ 2 No

b. If Yes, please list the gear and fill out table:
Gear:_____

	On a good day	On an average day	On a bad day	Split between how many?
Fish species generally caught				
Approx no/weight				
Approx income (TSh)				

H5. What other work do YOU personally have or have done in the last 5 years?

Occupation	Main job (Y/N)	If stopped, why stop?	Could get similar work now? (Y/N)	Money brings to household/week (Sh T)	Prefer to fishing? (Y/N)

H6. What activities do you and other people in your household do that bring in food or money to your house?					
Activity	Tick if respondent	No. of people (include respondent)	Rank of importance	Done for income or subsistence	Weekly income to household (Sh T)
Fishing					
Gleaning					
Mariculture					
Marketing marine products					
Farming					
Cash crops					
Livestock					
Small business					
Informal wage					

I. ASSETS/CAPITAL

I1. House material: Observe the houses/ or ask the respondent about the houses which belong to his/her households-confirm which is the households' main house, observe and note

I1a. House walls		
	Score	Number of houses
Thatch bad condition(b)	1	
Thatch good condition (g)	2	
Mud (b)	3	
Mud (g)	4	
Stone/Mud bricks – part of house	5	
Stone/Mud brick – all of house	6	
Cement blocks- part	7	
Cement blocks – whole	8	
Plaster/Paint – part	9	
Plaster/Paint - whole	10	

I1b. Household roof		
	Score	Number of houses
No roof	0	
Grass (b)	1	
Grass (g)	2	
Thatch (b)	3	
Thatch (g)	4	
Tin (b)	5	
Tin (g)	6	
Tile (b)	7	
Tile (g)	8	

I2. In this household do people:

I2a. own private transport?		
	Score	Number
None	0	
Bicycle	1	
Motorbike	2	
Car	3	

I2b. own household items?		
	Score	Number
Mobile phone	1	
Electric fan	2	
TV	3	
Refrigerator	4	
Satellite dish	6	

I3. In this household:

I3a. how do you access power?	
	Score
None	0
Battery	1
Generator/ Solar	2

I3b. how do you access water?	
	Score
River/Well/Pump free access	1
Buy water	2
Own private water	3
Have own tap outside	4
Have own tap in house	5
Have water tank	6

I3c. do you own house(s)?		
	Score	Number
Borrow house	1	
Rent house	2	
Own house	3	
Have title deed	4	

I4. Does the household own:

I4a. livestock?		
	Score	Number
None	0	
Chicken/Duck	1	
Goat/Sheep	2	
Cows	3	

I4b. farmland?		
	Score	Acres
Don't farm	0	
Borrow land	1	
Rent land	2	
Own land	3	
Have title deed	4	

J. QUESTIONS FOR FISHER WIVES

J1. Who owns the land on which you live?	☐ 1 Husband ☐ 2 Both ☐ 3 Another HH male	☐ 4 Me ☐ 5 Rented ☐ 6 Other
J2. Respondent's highest level of education?	☐ 1 None ☐ 2 Primary ☐ 3 Secondary ☐ 4 High school	☐ 5 College/University ☐ 6 Technical ☐ 7 Other (specify) _____
J3. Does your husband give you any money?	☐ 1 Yes	☐ 2 No

J4. If Yes, how much money does you husband give you in a:

- **a.** good week
- **b.** average **week**
- **c.** bad week

d. How often do you this receive money from your husband?

- ☐ 1 Daily
- ☐ 2 Few times a week
- ☐ 3 Once a week
- ☐ 4 Less than once a week

e. What do you spend this income on?

J5. Who in your household is responsible for the buying of food?

☐ 1 Me
☐ 2 Husband
☐ 3 Someone else

J6. Do you earn your own money?

☐ 1 Yes
☐ 2 No [Go to J8]

J7. From what activities do you earn money?

b. What you do with this money?
for example you give anyone / you reserve for your own / buy the needs of others?

J8. Do you account for any products brought into household? (E.g. food from gleaning/farming activities)

J9. Do you farm? ☐ 1 Yes ☐ 2 No

b. Do you farm for food or money?

☐ 1 **Money**
☐ 2 **Food** [Go to J9d]
☐ 3 **Both**

c. Where do you spend the money? (tick all that apply)

☐ 1 **Husband**
☐ 2 **Home**
☐ 3 **Self**

d. What products do you farm?

e. Who owns your house and garden?

f. Who owns your farmland?

J10. What activities do you and other people in your household do that bring in food or money to your house?

Activities:	Importance

J11. What level of control would you say you have over household income?

☐ **1** No control ☐ **2** A little control ☐ **3** Control equally with husband ☐ **4** A lot of control ☐ **5** Total control

B2 Facilitating legal framework in Tanzania

Policy/legislation	*Description*
National Land Policy, 1995.	Dual land tenure system. All land owned by State. Stipulates that all beaches are public.
Land Act, 1998.	Statutory right of occupancy stipulates the right to use and occupy land through Title Deed. Enables transfer of ownership of land
Village Land Act, 1998.	Customary right of occupancy gives the right to use and occupy land through Certificate of Customary Land issued by village councils and registered at District Land Registry.
National Fisheries Sector Policy, 1997.	Allocation and utilisation of fisheries resources in favour of rural communities. Involvement of fisher communities in planning, developing and managing fisheries resources.
The Fisheries Act No.6, 1970	Provides for protection, conservation, development, regulation and control of fish, fish products, aquatic flora and fauna Regulations surrounding spawning protection and pollution of waters
The Fisheries Act, 2003.	Enables community-based management Establishment of Beach Management Units (and strengthening of Village Liaison Committees within marine Parks). Joint management agreement between central government and local communities and equitable benefit sharing. Exclusion rights afforded to community-based management units
National Forest Policy, 1998.	Community-based. Village creation and ownership of forests. Joint management agreement between central government and local communities and equitable benefit sharing.
National Environmental Policy, 1997.	Local authorities responsible for overseeing planning process and establishing local environmental policies and regulations.

Source: Francis & Bryceson (2000) & Harrison et al. (2010)

B3 Village effects Model

	MALE		FEMALE	
Variables	**Model 4**	**Model 5 Village fixed effects**	**Model 4**	**Model 5 Village fixed effects**
Alt_inc	0.239*** (0.037)	0.191*** (0.040)	0.245*** (0.045)	0.327*** (0.086)
Grp_memb	0.034 (0.213)	-0.115 (0.294)	0.672*** (0.253)	0.786** (0.377)
Dep_work	0.030 (0.057)	0.020 (0.066)	-0.203*** (0.045)	-0.252*** (0.052)
Rely_hardtime	-0.074 (0.047)	-0.069 (0.063)	0.044 (0.099)	0.029 (0.094)
Avetrust	0.247*** (0.056)	0.321* (0.173)	0.241 (0.180)	-1.801 (1.354)
InTrust	-0.126 (0.172)	-	-0.299 (0.317)	-
Inpark	1.056 (0.660)	-	1.136 (1.182)	-
Age	-0.008 (0.007)	-0.010 (0.008)	-2.0e-04 (0.003)	-0.004 (0.003)
Education	-0.168 (0.160)	-0.090 (0.182)	0.328* (0.177)	0.359* (0.202)
HHsize	0.069 (0.054)	0.092* (0.055)	-0.016 (0.019)	-0.002 (0.024)
MSL	0.071 (0.046)	0.076* (0.045)	-0.087 (0.066)	-0.114 (0.088)
Land_area	-0.029* (0.016)	-0.016 (0.023)	-0.033 (0.044)	-0.022 (0.039)
Fish_income	0.086* (0.047)	0.084 (0.059)	-0.192 (0.320)	-0.451 (0.392)
Own_boat	-0.416** (0.472)	-0.481*** (0.173)	-	-
Dhow	-0.321*** (0.121)	-0.402*** (0.144)	-	-
Legal	-0.193 (0.203)	-0.152 (0.238)	-	-
Perceived_change	-0.285** (0.131)	-0.288** (0.141)	-0.081 (0.156)	-0.038 (0.161)
Better_off	-0.119 (0.117)	0.146 (0.117)	-0.234*** (0.054)	-0.232*** (0.056)
Cons_benefit	0.187*** (0.052)	0.209*** (0.052)	0.184** (0.074)	0.294*** (0.084)
Happy_child	-0.153** (0.077)	-0.167* (0.087)	-0.209*** (0.054)	-0.249*** (0.063)

	MALE		FEMALE	
Variables	**Model 4**	**Model 5 Village fixed effects**	**Model 4**	**Model 5 Village fixed effects**
Village				
Mkubiru		2.196 (0.976)**		-8.532* (5.014)
Mngoji		1.171* (0.979)		-9.500* (5.037)
Msimbati		-1.585** (0.731)		-4.382 (5.051)
Naumbu		-0.133 (0.869)		-9.934* (5.426)
Pemba		0.480 (1.218)		-7.660 (5.496)
Vill#avetrust				
Mkubiru		-0.422** (0.212)		1.976 (1.214)
Mngoji		-0.512 (0.228)**		1.982* (1.200)
Msimbati		0.554*** (0.160)		0.959 (1.212)
Naumbu		-0.059 (0.189)		2.330* (1.311)
Pemba		-0.196 (0.296)		1.623 (1.339)
_cons	-0.736 (1.001)	-0.829 (1.495)	-0.890 (1.271)	9.884* (5.237)
N	223	223	286	286
LogLikelihood	-116.439	-112.103	-147.787	-138.964
PseudoR^2	0.2454	0.2735	0.1589	0.2091

B4 Chow test: Male vs. female fishers

Variables	Pooled model	Interaction model:	Main effect	Interaction w.male
MALE	-0.444***		-1.641 [46]	
	(0.138)		(1.142)	
Alt_inc	0.251***		0.245***	-0.006
	(0.041)		(0.059)	(0.068)
Grp_memb	0.425***		0.672***	0.628*
	(0.143)		(0.253)	(0.337)
Dep_work	-0.141***		-0.203***	0.233***
	(0.032)		(0.045)	(0.064)
Rely_hardtime	-0.019		0.044	-0.117
	(0.047)		(0.099)	(0.124)
Avetrust	0.285***		0.241 [47]	0.006
	(0.096)		(0.180)	(0.167)
InTrust	-0.315**		-0.299	0.173
	(0.139)		(0.317)	(0.390)
Inpark	1.451***		1.056 [48]	0.080
	(0.560)		(0.660)	(1.329)
Age	-0.003		-2.02e-04	-0.008
	(0.004)		(0.003)	(0.007)
Education	0.075		-0.328*	-0.496***
	(0.101)		(0.177)	(0.191)
HHsize	0.023		-0.016	0.085
	(0.025)		(0.019)	(0.063)
MSL	-0.004		-0.087	0.158**
	(0.041)		(0.066)	(0.077)
Land_area	-0.051***		-0.033	0.004
	(0.016)		(0.044)	(0.054)
Fish_income	0.478		-0.133	0.218
	(0.519)		(0.221)	(0.223)
Own_boat	-0.401**		-0.416**	
	(0.188)		(0.177)	
Dhow	-0.255**		-0.321***	
	(0.108)		(0.121)	
Legal	-0.218		-0.193	
	(0.190)		(0.203)	
Perceived_change	-0.159		-0.081	-0.204
	(0.104)		(0.156)	(0.218)
Better_off	-0.165**		-0.234***	0.116
	(0.067)		(0.054)	(0.102)
Cons_benefit	0.216***		0.185**	0.002
	(0.012)		(0.074)	(0.120)
Happy_child	-0.198***		-0.209***	0.056
	(0.051)		(0.059)	(0.093)
_cons	-0.125		0.890	
	(0.735)		(1.271)	
N	509		509	
LogLikelihood	-274.027		-264.226	
PseudoR^2	0.2004		0.2290	
chi2 (5)			39.43	
Prob>chi2			0.0000	

[46] In line with pooled results (men & women) (p=0.151) given high correlation between male and interactions within Chow test. [47]Again, in line with pooled results (p=0.181). [48] Just above 10% at p=0.109. Significance levels in interaction model give results of t-tests across individual variables for main and interaction effect.

B5 Full model specifications: Varying Trust variables

Variables	Model 4 Average Trust	Model 6 Trust in Authority	Model 7 Trust in fishers from village	Model 8 Trust in fishers from other villages	Model 6b Trust in Authority marginal effects
Alt_inc	0.239***	0.243***	0.288***	0.359***	0.097
	(0.037)	(0.047)	(0.034)	(0.094)	(0.019)
Grp_memb	0.034	0.048	0.092	0.123	0.019
	(0.213)	(0.213)	(0.231)	(0.231)	(0.085)
Dep_work	0.030	0.048	-0.007	-0.021	0.019
	(0.057)	(0.060)	(0.068)	(0.084)	(0.024)
Rely_hardtime	-0.074	-0.080*	-0.082*	-0.083*	-0.032
	(0.047)	(0.045)	(0.047)	(0.048)	(0.018)
Avetrust	0.247***	0.103*	0.376***	0.110	0.041
	(0.056)	(0.058)	(0.143)	(0.166)	(0.023)
Trust*Inpark	-0.126	-0.104	-0.251	-0.155	-0.041
	(0.172)	(0.116)	(0.219)	(0.184)	(0.046)
Inpark	1.056	0.955**	1.706	1.131**	0.337
	(0.660)	(0.439)	(1.039)	(0.574)	(0.156)
Age	-0.008	-0.008	-0.010	-0.014	-0.003
	(0.007)	(0.007)	(0.007)	(0.011)	(0.003)
Education	-0.168	-0.177	-0.162	-0.087	-0.070
	(0.160)	(0.165)	(0.172)	(0.235)	(0.066)
HHsize	0.069	0.071	0.058	0.071	0.028
	(0.054)	(0.054)	(0.048)	(0.059)	(0.022)
MSL	0.071	0.064	0.091**	0.085**	0.025
	(0.046)	(0.049)	(0.044)	(0.042)	(0.020)
Land_area	-0.029*	-0.028*	-0.031	0.002	-0.011
	(0.016)	(0.016)	(0.020)	(0.031)	(0.051)
Fish_income	0.086*	0.087*	0.070*	0.093	-0.035
	(0.047)	(0.049)	(0.040)	(0.062)	(0.019)
Own_boat	-0.416**	-0.432**	-0.413**	-0.430**	-0.172
	(0.472)	(0.181)	(0.188)	(0.193)	(0.071)
Dhow	-0.321***	-0.316***	-0.377***	-0.327*	-0.125
	(0.121)	(0.114)	(0.146)	(0.186)	(0.006)
Legal	-0.193	-0.194	-0.207	-0.193	-0.077
	(0.203)	(0.199)	(0.192)	(0.222)	(0.079)
Perceived_change	-0.285**	-0.293**	-0.288**	-0.274*	-0.117
	(0.131)	(0.129)	(0.132)	(0.161)	(0.051)
Better_off	-0.119	-0.110	-0.140	-0.184	-0.044
	(0.117)	(0.129)	(0.121)	(0.120)	(0.046)
Cons_benefit	0.187***	0.193***	0.196***	0.210	0.077
	(0.052)	(0.057)	(0.085)	(0.059)	(0.023)
Happy_child	-0.153**	-0.150*	-0.180**	-0.192**	-0.060
	(0.077)	(0.080)	(0.034)	(0.076)	(0.032)
_cons	-0.736	-0.182	-1.436	-0.085	
	(1.001)	(0.940)	(0.938)	(1.271)	
N	223	223	221	199	
LogLikelihood	-116.439	-116.754	-113.812	-99.938	
PseudoR^2	0.2454	0.2433	0.2561	0.2748	

B6 Full model specifications: CE Trust

	Base model: nested			*Base model: conditional*		
	Coeff		SD	Coeff		SD
Closure	-0.017	**	0.009	-0.022	**	0.010
Net_small	0.268		0.394	0.315		0.358
Net_large	-0.852	*	0.455	-0.989	**	0.490
Payment_US	0.041	*	0.024	0.040		0.032
ASC	0.549		0.444	0.643	*	0.353
Avetrust_closure	0.002		0.002	0.003		0.003
Avetrust_netsm	-0.053		0.104	-0.055		0.095
Avetrust_netlg	0.075		0.118	0.071		0.129
Avetrust_payment	0.005		0.007	0.009		0.009
Avetrust_ASC	0.060		0.116	0.088		0.093
Log-L	-1597.0477			-1600.8923		
Adj-Pseudo R2				0.1350		
Waldchi	64.73					
Prob >chi	0.0000			0.0000		
N (choices)	5052			5054		
N(cases)	1684					
LR test for IIA P>chi2	-					