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Conservation payments in data-poor, developing-world fisheries

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A dissertation submitted for the degree of Doctor of Philosophy

Imperial College London

2016

Declaration of originality

This dissertation results entirely from my own work, and includes nothing that is the outcome of work done by or in collaboration with others except where this has been specifically indicated in the text.

Annabelle Bladon, May 2016

Copyright Declaration

Abstract

Effective fisheries management is limited in the developing world by weak institutions, inadequate financing and a lack of reliable data. Conservation payments are a novel concept in fisheries management. In this thesis I take a multidisciplinary approach to explore whether they could help to address gaps in traditional fisheries management, using the Bangladesh hilsa (*Tenualosa ilisha*) fishery and its ongoing payment scheme as a case study.

I develop a qualitative frame of reference against which current or potential hilsa management interventions could be evaluated, demonstrating that – even in data-limited fisheries – counterfactuals can be developed and used to guide management. In the absence of data for formal evaluation, I then investigate the scope for additionality in hilsa management by assessing the evidence for and against a reconstructed theory of change. Although the potential for overall additionality is equivocal, my findings demonstrate scope for individual elements of the management package to have had additionality, and provide some support for the use of conservation payments.

As is common in artisanal fisheries management, hilsa management is focused on the protection of juveniles. Through population modelling, I demonstrate that size selectivity is much less important than catch volume, in terms of effect on overall hilsa population biomass. This analysis suggests that the targeting of payments would benefit from a more rigorous ecological foundation. Through statistical modelling of household survey data, I find a strong spatial pattern in payment distribution that reflects the political economy of Bangladesh rather than the official social goals of the scheme. I also find evidence of strong trade-offs between social and ecological goals.

Finally, I investigate the potential for Conservation Trust Funds (CTFs) to enhance the sustainability of payments. Developing-world fisheries pose challenges to the translation of conservation payments from concept to reality. I find that CTFs can support and catalyse the development of enabling conditions for sustainable payment institutions, but only if best practice standards are followed.

Acknowledgements

Grateful thanks go first to my supervisors: E.J. Milner-Gulland, for taking a chance on me and for such patient dedication; Katherine Short, for her ideas, enthusiasm, and for showing me how networking is really done; and Essam Yassin Mohammed, for bringing me on board the hilsa project and giving me such a valuable insight into the world of policy research.

My fellow ICCS members have also been a source of unwavering support; I could not have asked for a more inspiring and merry bunch with whom to spend my Silwood days. A special mention should go to CLSC, whose email threads will be dearly treasured (but never shared). And forever more, when I get in a muddle, I shall simply ask myself, 'What would Joe Bull do?'

In London, I thank IIED's Sustainable Markets Group, who were kind enough to provide me with a desk for two years. In Bangladesh, BCAS provided the logistical support and data collection that made this research possible. In particular, I am grateful to Liaquat Ali, Belayet Hossain, Golam Kibria, Olena Riza and Mizanur Rahman. My thanks also go to Anne Lillelund, who provided me with a home from home, while always watching out for my road safety and nutritional needs. In New York, I am indebted to NYU's Department of Environmental Studies and, particularly, to Jennifer Jacquet, who made it possible for me to finish my thesis without spending my final year on an aeroplane.

For technical advice, my thanks go to Genevieve Finerty, Francois Husson, Henry Travers, Emilie Beauchamp, Aidan Keane and Oliver Burdekin. For valuable guidance at various stages, I thank Julia Blanchard, David Agnew and Rashid Sumaila. For constructive feedback on specific chapters, I am grateful to Ben Thompson, Nadia Richman, Lorenzo Rosenzweig Pascuel, María José González, Maurice Brownjohn, Domingos Gove, my anonymous reviewers, and Mark Wheeler – who I wish I had asked to proof every chapter. For patiently answering questions on the complexities of fisheries management in Bangladesh, I thank Masud Ara Mome and Monirul Islam. And of course, I am grateful to everyone who gave up their time to be interviewed for this research.

Finally, my most sincere thanks go to my friends and family for putting up with me throughout this seemingly never-ending education: Mum and Dad, for having faith in me and for sending me to that terrifying piano teacher, Mrs Bather, all those years ago, who showed me the value of hard work; Jess, for letting me move in with her boyfriend before she did; James, for his generosity and outstanding cuisine (let's hope this means I'm gone for good); and Harry, for understanding, for encouraging my curiosity, and for never taking any nonsense.

This research was funded by a studentship from the Biotechnology and Biological Sciences Research Council, with additional support from the Darwin Initiative and the Department for International Development, via the International Institute for Environment and Development.

BBSRC ICCS Interdisciplinary Centre for Conservation Science DARWIN INITIATIVE iied International Institute for Environment and Development

NYU

“মাছে-ভাতে বাঙালী” (Fish and rice make a Bengali)

Bengali proverb

Abbreviations and acronyms

AIC	Akaike's Information Criterion
BACoMaB	Banc d'Arguin & Coastal & Marine Biodiversity Trust Fund
BDT	Bangladesh Taka (currency)
BFDC	Bangladesh Fisheries Development Corporation
BFRI	Bangladesh Fisheries Research Institute
BLUPs	Best Linear Unbiased Predictors
BNP	Bangladesh Nationalist Party
BoB	Bay of Bengal
BOBLME	Bay of Bengal Large Marine Ecosystem
CBF	Caribbean Biodiversity Fund
CPR	Common Pool Resource
CPUE	Catch Per Unit Effort
CTF	Conservation Trust Fund
df	degrees of freedom
DoF	Department of Fisheries
EBFM	Ecosystem-Based Fisheries Management
EEZ	Exclusive Economic Zone
EU	European Union
FAMD	Factor Analysis for Mixed Data
FIP	Fisheries Improvement Project
FMCN	Mexican Nature Conservation Fund
GDP	Gross Domestic Product
GLMM	Generalised Linear Mixed Effects Models
HFMAP	Hilsa Fisheries Management Action Plan
IIED	International Institute for Environment and Development
IPCC	International Panel on Climate Change
ITQ	Individual Transferable Quota
IUU	Illegal, Unreported and Unregulated

KI	Key Informant
KII	Key Informant Interview
LHP	Life History Parameter
LMM	Linear Mixed Effects Models
MAR	Mesoamerican Reef
MoFL	Ministry of Fisheries and Livestock
MPA	Marine Protected Area
MSC	Marine Stewardship Council
MSY	Maximum Sustainable Yield
NGO	Non-Governmental Organisation
PACT	Protected Areas Conservation Trust
PES	Payments for Ecosystem Services
PIPA	Phoenix Islands Protected Area
PNA	Parties to the Nauru Agreement
SD	Standard Deviation
SE	Standard Error
SR	Stock-Recruitment
TURFs	Territorial Use Rights for Fishing
USD	United States Dollars
VGF	Vulnerable Group Feeding programme
WCPO	Western and Central Pacific Ocean
YPR	Yield Per Recruit

A note on the text

Bengali words are written phonetically in italics. I have followed the spellings used by the Bangladesh Centre for Advanced Studies, but it should be noted that, since standards for the Romanisation of Bengali have not been adopted uniformly, phonetic spellings can vary widely.

Contents

List of figures

List of tables

Chapter 1

Introduction

1.1 Context and problem statement

Fish and other seafood play a vital role in supporting marine and coastal[1] ecosystems, which in turn provide their own multitude of services to humans (TEEB 2012; Liquete et al. 2013). They also directly support human wellbeing through their contributions to employment and food security (Smith et al. 2010; Teh & Sumaila 2013; Béné et al. 2016). However, a global decline in marine fish stocks and ecosystem health has been observed over the last half of the 20th century (Myers & Worm 2003; Worm et al. 2006; FAO 2014). Through poor governance, the provision of harmful subsidies and subsequent overcapacity, a growth in demand for fish products has led to overexploitation (Sumaila 2012; Srinivasan et al. 2012).

Management tools such as gear restrictions, spatial closures, and capacity reduction have great potential to reverse this decline (Beddington et al. 2007), and many developed countries are now implementing them effectively (Hilborn 2007). As the goals and perspectives of fisheries science and marine conservation begin to converge (Salomon et al. 2011), there has also been a shift away from traditional single-species fisheries management towards an ecosystem-based approach (Ward et al. 2002; Jennings et al. 2014). Furthermore, in recognition of the strong economic argument for fishery reform (Dyck & Sumaila 2010; Costello et al. 2012; Sumaila et al. 2012), marine resources and ecosystems play an increasingly prominent role in global political commitments to sustainable development (Veitch et al. 2012; UN 2015a).

Nevertheless, current efforts are not operating at the scale necessary to rebuild fisheries globally (Mora et al. 2009; Worm et al. 2009). Effective implementation of sustainable management is limited by a lack of will and ability to bear the inevitable short-term costs (Shertzer & Prager 2007; Ye et al. 2012; Rangeley & Davies 2012), particularly in small-scale

[1] Hereafter referred to as marine.

and artisanal[2] developing-world fisheries (Carbonetti et al. 2014; Brown et al. 2015; Ovando et al. 2016). Often dispersed in remote locations with complex social structures, these fisheries tend to be poorly assessed, if at all, and neglected by national fisheries policies. If management is implemented, the personal costs may be perceived by individual fishers or groups of fishers to exceed the benefits, which may lead to resistance and non-compliance, particularly when enforcement is weak (Wells 1992; McClanahan et al. 2012; Wallace et al. 2015). These costs and benefits may be social as well as economic; individuals might feel that regulations threaten their rights or access to resources, or that they are inequitable (McDermott et al. 2013). It is, therefore, critical that developing-world fisheries management evolves beyond the conventional command-and-control model to recognise fisheries as complex and dynamic social-ecological systems (Berkes 2012; Leenhardt et al. 2015; Purcell & Pomeroy 2015), in which the consideration of incentives is crucial to achieving behavioural change (Castilla & Defeo 2005; Hilborn et al. 2005; Begossi 2014).

Since the vast majority of people engaged in fishing are part of small-scale and artisanal fisheries in the developing world (World Bank 2010; Barnes-Mauthe et al. 2013), the potential effects of fishery reform on poverty alleviation could be substantial (Garcia & Rosenberg 2010; Begossi 2013; Wilen 2013). Moreover, despite the management challenges that they present, small-scale fisheries are potentially more sustainable than industrial fisheries; they tend to use less energy-intensive gears, fish closer to shore, have lower discard rates, and use less fuel to catch the same amount of edible fish (Jacquet & Pauly 2008; de Melo Alves Damasio et al. 2016).

[2] Definitions of 'small-scale' and 'artisanal' commonly depend on location, and are often used interchangeably, but small-scale tends to refer to the size of a fishery or vessel, and artisanal to the relative level of technology. Artisanal fisheries are defined by the FAO as 'traditional fisheries involving households (as opposed to commercial companies), using relatively small amount of capital and energy, relatively small vessels (if any), making short trips, close to shore, mainly for local consumption' (Garcia 2009). In practice, they can be subsistence or commercial, and provide for local consumption or export.

In line with the recent surge in 'market-based' approaches to resource management and conservation, a range of economic instruments[3] are becoming available for fisheries management (Innes et al. 2015). By attempting to realign the incentives faced by individuals with the objectives of management, they are advocated for their potential to alleviate some of the short-term costs of fishery reform (Ferraro & Gjertsen 2009; Davies & Rangeley 2010; Fujita et al. 2013). Conservation payments, which include compensation and positive incentives for actions taken to maintain or enhance a resource, are a group of economic instruments that are well established in terrestrial conservation (Vatn 2010; Muradian & Rival 2012). Compensation can be distinguished from incentives on the basis that compensation payments aim to offset costs, whereas incentives aim to change behaviour voluntarily, and may even have additional benefits (Vatn 2010). But a payment can both compensate and incentivise, and I focus here on the incentive element. Payments for ecosystem services (PES) is, in theory, a subset of incentive-based approaches – although it is commonly used as blanket term for a whole range of types of payments. Conceived as a market solution to environmental externalities (Wunder 2005; Engel et al. 2008), PES can more usefully be viewed as an instrument aiming to improve the provision of ES by resource users to beneficiaries, by offering conditional positive incentives for behavioural change (Sommerville et al. 2009; Muradian 2013). Since ES are usually common pool or public goods, I follow Muradian's (2013) view that their management is more of a social dilemma than an externality problem. Although PES should not, in theory, pay individuals for already legally enforceable behaviours, in some circumstances a 'carrot-and-stick' approach can be used to strengthen individual and collective motivations for compliance, enabling poor communities to reduce their reliance on resources under protection while facilitating a more

[3] I do not use the umbrella term 'market-based' in this thesis, because although all of these instruments rely on price signals, many do not rely on markets at all, and the term places a restrictive focus on efficiency (Pirard 2012; Boisvert et al. 2013; Muradian & Gómez-Baggethun 2013; Hahn et al. 2015; Vatn 2015).

equitable distribution of the costs of doing so (Wunder 2007; Kosoy et al. 2008; Clements et al. 2010; Sommerville et al. 2010a; Gross-Camp et al. 2012).

Documentation of the implementation of conservation payments, particularly PES, in marine environments is still relatively rare, as is critical discussion of their social and ecological impacts (Begossi et al. 2011a; Lau 2012; Mohammed & Wahab 2013; Binet et al. 2013). Empirical research to date has focused mostly on the use of compensation in small-scale, artisanal fishing communities around Marine Protected Areas in Brazil, Kenya, the Coral Triangle, and Bangladesh (e.g. Clifton 2013; Islam et al. 2014; Begossi 2014; Corrêa et al. 2014), but much of this is hypothetical (Barr & Mourato 2009; Barr 2012). There is, however, an opportunity to transfer PES experience from terrestrial conservation to fisheries. This could support the use of conservation payments as an alternative or complementary approach to conventional regulatory fisheries management, thereby helping to reconcile the conservation of fishery resources with sustainable development and poverty alleviation.

1.2 Aim and objectives

The aim of this thesis is to explore the potential for conservation payments to meet social and ecological objectives in data-deficient, developing-world fisheries. I approach this using the management of hilsa shad (*Tenualosa ilisha*) in Bangladesh as a case study. Hilsa are in the clupeid family, with herrings and sardines, and are found throughout South and Southeast Asia in marine and freshwater. Not only are they a flagship species for Bangladesh, a symbol of national pride, but up to 500,000 people directly depend upon the species for their livelihoods, particularly in coastal communities (Islam et al. 2016). The fishery is typical of small-scale, developing-world fisheries; it is complex, poorly understood, and characterised by centralised governance and weak monitoring and enforcement. It does, however, present a rare carrot-and-stick management approach, and thus a real-world platform for the exploration of the feasibility and potential implications of such an approach in challenging circumstances.

This thesis will address the aim through the following objectives:

1. Explore the suitability of and challenges to the design and implementation of PES in a developing-world fisheries context;
2. Develop a frame of reference for the Bangladesh hilsa fishery – comprising ecological, institutional, social, economic and physical components – against which current or potential conservation payment schemes could be evaluated;
3. Identify which fishing households have the greatest ecological impact on the hilsa fishery, in terms of their socioeconomic characteristics and fishing activities;
4. Explore whether the current hilsa management package, including the compensation scheme, has had any additional impact on the sustainability of the fishery to date, and whether it has scope for additionality in the future;
5. Assess whether the compensation for hilsa fishers is reaching the 'right' people, in terms of the scheme's social and ecological objectives;
6. Explore whether Conservation Trust Funds can provide a sustainable framework for conservation payments in developing-world fisheries, with a focus on the Bangladesh hilsa fishery;
7. Make recommendations for the development and improvement of the compensation scheme for hilsa fishers in Bangladesh and for payment schemes more broadly, in a data-poor, developing-world fisheries context.

1.3 Thesis outline

Subsequent to this introductory chapter, the thesis is structured as follows (Fig. 1.1):

Chapter 2: Payments for Ecosystem Services in developing-world fisheries

Here I consider the potential role of marine PES in addressing current challenges and gaps in fisheries management. I do this through a comprehensive review of the literature and a conceptual analysis of four contrasting real developing-world case studies – one of which is the main case study used throughout the thesis, the Bangladesh hilsa fishery.

A version of this chapter has been published as:

Bladon, A.J., Short, K.M., Mohammed, E.Y. and Milner-Gulland, E.J. (2014). Payments for ecosystem services in developing world fisheries. *Fish and Fisheries.* DOI: 10.1111/faf.12095

Chapter 3: Creating a frame of reference for hilsa conservation and management interventions in Bangladesh

This chapter describes the social-ecological system of the case study in depth and thus provides a basis for subsequent chapters. I combine qualitative and some quantitative analyses of secondary datasets and literature in a qualitative way to explore not only ecological trends in the hilsa fishery, but also patterns of social, economic, institutional, and physical change relevant to its management. I show that useful counterfactuals can be developed to guide fisheries management even in complex, data-deficient systems.

Chapter 4: Characterising the impacts of selective fishing on the Bangladesh hilsa population

Here I use minimal life history parameter data to model the potential relative ecological impacts of size selectivity on hilsa population biomass. I show that the focus of hilsa fishery management on the protection of juveniles may not be the most effective approach. Having characterised the hilsa fishery using household survey data, I then assess the relative potential impacts of different harvesting regimes at the household level, using reported size-selectivity and average hilsa catch volumes to develop an index. Statistical modelling shows this household-level index of ecological impact to be influenced largely by spatial factors, and to a lesser extent by boat ownership and fishing location, which appear to be driven largely by catch volume rather than selectivity.

Chapter 5: Exploring the ecological additionality of hilsa fishery management in Bangladesh

Building on the frame of reference, I reconstruct a *post-hoc* theory of change for hilsa management, before assessing the available evidence (literature and primary and secondary data) behind each of the assumptions underpinning its logical pathways. Overall, I find that the potential for additionality (i.e., the degree to which a management regime, or component thereof, improves the status of the fishery over and above how it would be in the absence of that management) is very uncertain. But although the use of payments in this fishery is not supported by robust social or ecological evidence of success, I conclude that in the absence of effective top-down enforcement, development and improvement of the compensation scheme is necessary. This demonstrates that even when data deficiencies limit rigorous impact evaluations, potentially useful studies of the scope for, or confidence in, additionality can still be conducted.

Chapter 6: Does compensation for hilsa fishers in Bangladesh reach the 'right' people?

The compensation scheme is designed to provide food support to the poorest and most vulnerable households targeting juvenile hilsa and living inside and around sanctuary areas, during a period when fishing is banned. I present data from a household survey of compensation recipients and non-recipients and use statistical modelling to profile hilsa fishers who catch juveniles, to identify the current correlates of compensation allocation, and to explore perceptions of fairness in allocation. I find that the pattern of compensation allocation is largely spatial rather than based on relevant household characteristics (e.g. indicators of poverty, vulnerability, or fishing activities), and largely perceived to be unfair. I then demonstrate how the spatial distribution of compensation would change under alternative targeting scenarios that might improve the efficiency and effectiveness of the scheme, such as targeting those who are most dependent on fishing for their livelihood. I provide evidence not only of the need for a more focused and transparent targeting strategy in Bangladesh, but of the need for improved targeting effectiveness, which highlights a challenge for developing-world payment schemes to achieving social objectives.

Chapter 7: Can Conservation Trust Funds provide a sustainable framework for conservation payments in developing-world fisheries?

The Government of Bangladesh is currently developing a Conservation Trust Fund (CTF) to support the management of the hilsa fishery. Using a literature review and key informant interviews, I demonstrate how CTFs can potentially, under best practice, support the development or improvement of conditions that enable sustainability in developing-world fisheries conservation payments. This will, however, depend on best practice design and implementation; in Bangladesh, the institutional context may limit potential for a CTF to support improvements in the sustainability of the compensation scheme for hilsa fishers. I provide recommendations for addressing the key challenges that a CTF may face to providing a sustainable framework for hilsa conservation payments.

Parts of this chapter are published in:

Bladon, A.J., Mohammed, E.Y. and Milner-Gulland, E.J. (2014). A Review of Conservation Trust Funds for Sustainable Marine Resources Management: Conditions for Success. IIED Working Paper. IIED, London

Chapter 8: Discussion.

This chapter provides a synthesis of research findings, key implications for fisheries management and conservation science, policy recommendations, and directions for future research.

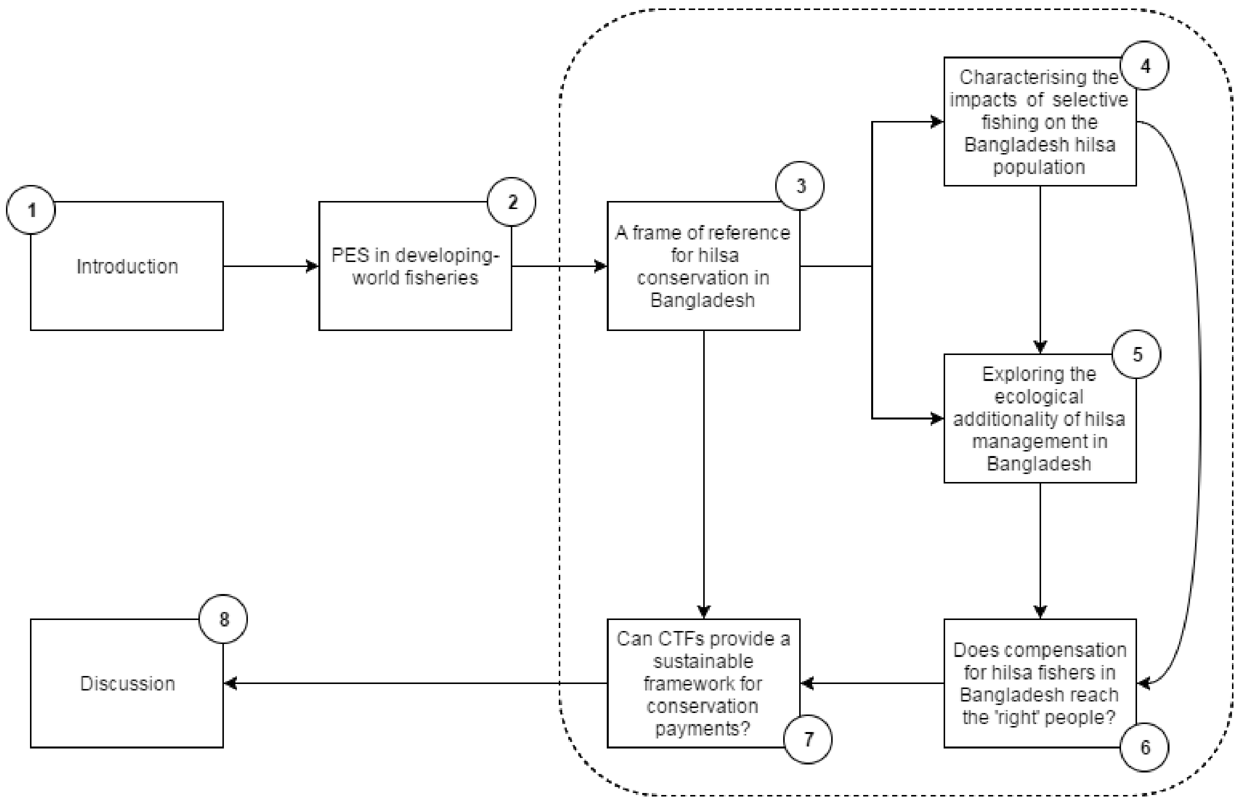


Figure 1.1: Conceptual framework for thesis. Boxes are numbered by chapter and arrows indicate logical flow of chapters. Dotted line indicates which chapters focus on the case study. PES = Payments for Ecosystem Services; CTFs = Conservation Trust Funds.

Chapter 2

Payments for ecosystem services in developing-world fisheries

2.1 Introduction

Payments for Ecosystem Services (PES) is a tool widely used in terrestrial conservation to change incentives for environmental decisions (Sattler & Matzdorf 2013). Traditionally viewed in terms of Coasean economics as a market approach to the internalisation of environmental externalities (Wunder 2005; Engel et al. 2008), in practice numerous different institutional forms of PES exist, few of which conform to pure market transactions (Muradian et al. 2010; Farley & Costanza 2010; Muradian 2013a; Sattler & Matzdorf 2013).

Although there has been limited empirical analysis of PES implementation and efficacy (Farley & Costanza 2010), evidence suggests that PES can effectively complement conventional regulatory approaches to conservation, both in developed and developing countries (Wunder et al. 2008). Often promoted as a 'win-win' approach to achieving environmental and social impacts, PES is historically most popular in middle-income developing countries (Pagiola et al. 2005; Grieg-Gran et al. 2005; Bulte et al. 2008). While any conservation intervention is less likely to succeed in the weak institutional settings that are common in developing countries, PES may facilitate the strengthening of institutions and ease cooperation in cases where governance is poor (Kosoy et al. 2007; Clements et al. 2010; Wunder 2013). Furthermore, although the social impacts of PES have not been well measured, there is some evidence that it can enhance rural livelihoods (Clements & Milner-Gulland 2014; Ingram et al. 2014; Bremer et al. 2014). The approach can also be sustainable in the long term, especially when integrated with other instruments such as co-management (Chapter 7; Clements et al. 2010; Fisher 2012; Goldman-Benner et al. 2012; Sarkki & Karjalainen 2015), and thus can be an innovative way to generate sustainable financing for conservation (Sattler & Matzdorf 2013; Bos et al. 2015).

Marine conservation is increasingly adopting an ES framework, and a range of economic instruments and market-based approaches are becoming available (Fujita et al. 2013). However, the concept of marine PES is still nascent, largely due to the fluid, transboundary and often common pool nature of marine ecosystems. Discussion has been directed towards the development of payments for the protection of coastal ecosystems such as mangroves, particularly for their role in carbon sequestration, where lessons can be drawn more easily from terrestrial experience (Lau 2012; Locatelli et al. 2014). In recent years, PES has received attention as an approach that could complement conventional regulatory fisheries management, particularly in small-scale, developing-world fisheries (Pagiola 2008; Lau 2012; Mohammed & Wahab 2013; Binet et al. 2013; Micheli et al. 2014; Innes et al. 2015). Furthermore, some authors hypothesise that by linking marine ES buyers with providers, PES could provide a novel way to guide investment in sustainable fisheries (Short 2012, 2014). However, critical discussion and empirical analysis are limited (Barr & Mourato 2009; Begossi et al. 2011a; Barr 2012; Hallwass et al. 2013; Begossi 2014; Corrêa et al. 2014).

In this chapter I explore the potential for a PES approach to help address current challenges and gaps in commercial fisheries management in the developing world. Although they may present the greatest tests for PES, regarding weak governance, ill-defined property rights, and lack of technical resources and capacity (Grafton et al. 2008), developing countries also present some of the greatest opportunities for additional gains, particularly through potential contributions to food security and poverty alleviation. I first outline the principles which, in theory, distinguish PES from other fisheries management tools and explore the ways in which PES might be applied in fisheries. I then discuss the institutional and environmental preconditions that would help to ensure successful delivery of a fisheries PES scheme in practice, before illustrating some real world opportunities for and limitations to the approach using four contrasting case studies.

2.2 The defining principles of PES in a fisheries context

PES is most commonly defined as a voluntary transaction whereby a well-defined ES (or actions likely to secure it) is 'bought' from at least one ES provider by at least one buyer, if and only if the payment is conditional on provision of that ES (Wunder 2005). In other words, it translates external values into positive incentives for behavioural changes that are expected to increase the provision of one or more ES. These positive incentives, which might be financial or in-kind, are paid to the ES provider (the individual or group accountable for ES delivery) for carrying out a specific management activity. Although Wunder's definition is the clearest and most detailed, in practice PES schemes rarely fulfil each of his five criteria (Muradian et al. 2010). Recent conceptualisations focus on the core principles of positive incentives and conditionality, and emphasise the consideration of additionality – the degree to which a PES scheme adds value over what would have happened in its absence (Sommerville et al. 2009; Tacconi 2012; Muradian 2013a). Here I outline the principles which theoretically set PES apart from other fisheries management tools, and the extent to which they may be addressed in a fisheries context (Fig. 2.1).

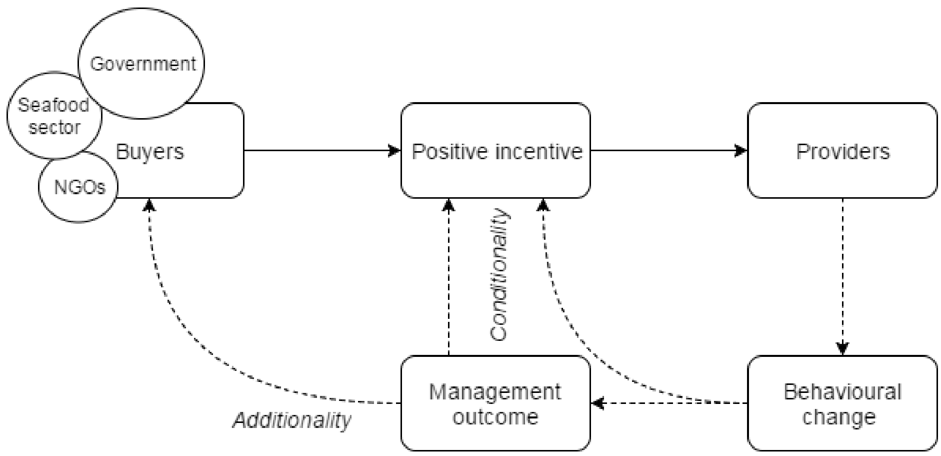


Figure 2.1: Schematic for a fisheries Payment for Ecosystem Services (PES) scheme. Buyers may include seafood supply chain actors, governments, and NGOs; management outcomes may include fishery recovery and therefore enhanced profitability or provision of other ES of concern to the buyer; solid arrows represent payment flow, which should be conditional on provider behaviour or management outcomes.

2.2.1 Clearly defined ES

Theory states that the service(s) in question should be clearly defined in order for provision to be quantifiable (Wunder 2005; Tacconi 2012), but the interconnectivity of marine ES makes them difficult to define in time and space (Sousa et al. 2016). For example, if payments were made to fishers for avoiding a specific area of habitat, with the aim of increasing fish stocks, this service might be difficult to disentangle from other ES that might also be delivered through this action (such as improved benthic habitat). 'Bundled' PES schemes could be used to address these challenges, whereby payment is received for multiple ES grouped together in a single package of conservation outcomes (e.g. habitat conservation or fishery performance), which, in turn, may raise the incentive for management activities with multiple potential outcomes and reduce trade-offs between ES (Lau 2012). Lau (2012) also considers the benefits of 'stacking', whereby separate payments are generated for distinct ES, but this kind of payment structure is less able to reflect the complexity and interconnected nature of ecosystems. Successful implementation of bundling in terrestrial conservation has been rare (Engel et al. 2008); it can be more costly and difficult than dealing with single ES due to the involvement of more stakeholder groups (Lau 2012). However, efforts are underway to address these challenges (Wendland et al. 2010; LaRocco & Deal 2011). In particular, lessons from work on markets for the bundling of agricultural products with other ES in Africa and Latin America may be applicable to the bundling of seafood with other ES (Andersson et al. 2010).

2.2.2 Buyers and providers

The potential providers in a fisheries PES scheme range from individuals or communities of small-scale fishers to industrial fishing fleets to nation states. If a provider is to receive payment and thereby be held accountable for ES delivery, or at least for management actions, they must have some level of ownership or control over ES delivery (Wunder 2013). However, property rights are often unclear in fisheries, particularly in the developing world where they may be traditional or undocumented, making the identification of providers potentially difficult (Muradian 2013b). Even when rights are defined and enforced, the high mobility of marine resources makes it difficult to prevent others from accessing the resources and their ES, which might reduce the suitability of a PES approach (Kemkes et al. 2010). Furthermore, fisheries stakeholders tend to be numerous, widely dispersed, and mobile. In a scheme where fishers act as providers, as a group they may have a strong incentive to change their behaviour, but as individuals they each have an incentive to avoid doing so (Pagiola 2008). This fragmentation could also lead to multiple and conflicting claims for payment and raise transaction costs – issues encountered in the management of forests, another system in which goods and services are supplied by a variety of different stakeholders at a range of geographic scales and in which land tenure is complex (Wunder 2007; To et al. 2012). There can also be interdependence between fisheries sectors: for example, shellfish gleaners might be affected by fisheries practices, but not necessarily included in a PES scheme.

The buyer in a PES scheme can be any actor who benefits from service provision – fishers, an NGO, a government body, a private company, consumers, or any combination of these. Interested buyers may vary with the ES or action that is targeted: a government might have an interest in paying for an overall management plan, and an NGO for actions such as reduction in the use of damaging gear, whereas a private company is more likely to be interested in a specific ES such as fish provision. Although no single source is likely to be sufficient, the seafood sector is a potentially significant and largely untapped source of investment for PES (Blasiak et al. 2014). It has already shown willingness to support the transition to sustainable fisheries through corporate social responsibility programmes and involvement in fisheries improvement projects (FIPs), certification schemes and ecolabelling (Micheli et al. 2014; Chaplin-Kramer et al.

2015). However, research suggests that this willingness is not fully exploited (Vallejo et al. 2009; Short 2011, 2012). PES could be used to capture this willingness and guide investment, thereby strengthening supply chain accountability. For example, by providing fishers with an economic incentive to avoid fishing in nursery grounds, private companies might invest in the natural capital of this habitat; i.e., the fish stocks and the flow of ES which support these stocks. Through increased fishery profitability, this mechanism could in turn stimulate additional public or private investment.

2.2.3 Voluntary transaction

It is generally agreed that PES should at least be voluntary for the provider, in order for the payment to have conditionality, but that this is less important for the buyer (Tacconi 2012; Lau 2012). For example, payments might be generated through taxation, where the buyers themselves may not necessarily be directly involved in the transaction (Goldman-Benner et al. 2012). Others propose that the extent to which the transaction is voluntary depends on institutional context; for instance, if payments are being used to alter behaviours which are already illegal, in combination with regulatory approaches, then the providers may not act voluntarily (Sommerville et al. 2009; Farley & Costanza 2010). Furthermore, in cases where communities as a whole are acting as the provider, as may often be the case in artisanal fisheries, the transaction could be voluntary at the level of the group rather than the individual (Sommerville et al. 2009).

2.2.4 Conditionality

Conditionality is the methodological core of PES, the component which creates a consequence for not providing the ES (Sommerville et al. 2009). Payments that are conditional on outcomes, i.e., ES provision, are thought to be the most effective (Ferraro & Kiss 2002; Banerjee et al.

2013), but in practice the technical and financial challenges of monitoring mean that provider compliance with the agreed management actions is often used instead (Sommerville et al. 2009, 2011; Pattanayak et al. 2010). Due to the high levels of uncertainty attached to most marine ES, conditionality on their provision may be particularly difficult to establish. Where the target is fish provision or habitat protection, conditionality on, for example, stock status or area protected may be relatively easy to maintain, but proxy indicators are much less available for regulating and cultural services than they are for provisioning ES (Liquete et al. 2013). Action-based payments still require enforcement, however, and may therefore still lead to high transaction costs in marine systems.

2.2.5 Additionality

Additionality is a measure of the outcome of an intervention relative to the situation in its absence, and thus an important indicator of the benefits and cost-effectiveness of PES as an approach (Chapter 5; Sommerville et al. 2009; Tacconi 2012). It is vital to assess cost-effectiveness, not only in relation to the original situation, but also to alternative management approaches. The measurement of additionality can help to avoid the allocation of payments to individuals who were already going to carry out conservation actions. It can likewise help to inform the prediction of when a scheme should end (continuing payments beyond the point when they are having additional impact would not be cost-effective); and it can be crucial in generating and maintaining financial support (Wunder et al. 2008; Pattanayak et al. 2010). However, additionality is often difficult to establish and therefore rarely explicitly monitored in PES schemes (Pagiola & Rios 2008; Honey-Rosés et al. 2009). The estimation of baselines and counterfactuals in the marine environment suffers from similar challenges to those encountered in the monitoring of marine ES provision, particularly in small-scale developing-world fisheries, which often lack even the most basic management tools such as stock assessments. It may be more practical to assess actions rather than outcomes in such cases (Muradian 2013a). Surveys assessing perceptions of compliance and scheme acceptability could also be used to monitor and evaluate some components of PES impact over time (Hallwass et al. 2013; Bennett 2016). Social diffusion, where the change in behaviour of a small percentage of fishers may have positive impacts on the behaviour of others, should be taken into account when assessing overall additionality of a scheme (Goldman-Benner et al. 2012), as should displacement, where payments allow ecosystem damage to be shifted elsewhere, resulting in no net change in fisheries practice or ecosystem condition on a broader scale (Wunder et al. 2008).

2.3 Scope for PES in fisheries

A fisheries PES scheme may be used to support an overall management plan, a specific management action or the provision of a specific ES. It might, for example, be a short-term measure to support seafood provision, or the adoption of actions known to increase seafood provision. However, the principles of an ecosystem approach would ideally require a broader focus which encompasses additional ES. A scheme supporting a management plan, conservation outcome, or bundle of ES is more likely to maintain additionality over alternative approaches, and therefore may be more sustainable in the long run.

The use of economic instruments, including PES, as tools to reduce the negative impacts of commercial fishing has been reviewed by Innes et al. (2015). There are examples in the peer-reviewed literature of proposed mechanisms that compensate fishers for earnings lost or costs incurred through a change in gear type, fishing location or fishing practice (Döring & Egelkraut 2007; Vinha et al. 2010), and of 'conservation agreements' whereby philanthropic investment is chanelled via NGOs to local communities in exchange for stewardship activities (Niesten et al. 2013). In particular, the feasibility of government and tourist compensation payments to small-scale artisanal fisheries has been investigated (Barr & Mourato 2009; Clifton 2013), but examples of actual implementation are rare (see Table 2.1). Furthermore, the extent to which many of these mechanisms conform to the defining principles of PES is unclear. Though they use payments, conditionality rarely appears to be in place. For example, the Brazilian Government operates a payment scheme called the *defeso*, whereby fishers receive compensation for costs incurred during periods of fishery closure. However, criticism of their systems of monitoring, control and surveillance (MCS) indicates that payments are not fully conditional on compliance, nor is the transaction voluntary from the perspective of the providers (Begossi et al. 2011a; Corrêa et al. 2014). According to Begossi et al. (2011a), improvements in the scheme could allow it to serve as the 'basis for more effective PES instruments'.

One of the only examples of a fisheries payment mechanism formally and legitimately referred to as PES in the literature to date is that of the Banc d'Arguin National Park in Mauritania, where the European Union (EU) allocates part of its payment for the EU-Mauritania Fisheries Partnership Agreement to the management of the park (Binet et al. 2013). By investing in the biomass productivity of the park, the EU is protecting nursery and breeding sites which contribute to the productivity of its commercial fishing grounds. This mechanism is the first international payment of its kind and undermines some of the theoretical barriers to marine PES. It has led to enhanced management of the park, and there is potential for the mechanism to be extended to benefit other major fisheries in Mauritania and in Guinea Bissau (Binet et al. 2013).

Seafood certification schemes are sometimes described as a form of PES (e.g. Forest Trends & The Katoomba Group 2010), though they fulfil only some of the defining principles. The most prevalent of these, the Marine Stewardship Council (MSC), aims to create positive market incentives for sustainable fishing by shifting consumer demand towards MSC-certified products, thereby generating a return on investment in sustainable practices. In theory, the certified fishery (voluntary provider) receives a price premium from the consumer (the buyer) which is conditional on the effective management and health of the fishery and its impact on the environment (Roheim et al. 2011; MSC 2013). Although the MSC has begun quantitatively to evaluate its performance (MSC 2013), it has been criticised for weak linkages between certification and conservation outcomes (Ward 2008; Jacquet et al. 2010; Christian et al. 2013; Micheli et al. 2014). Certification of bundles of marine ES may increase opportunities for ecological and socioeconomic impacts, but it does not yet explicitly address defined ES other than seafood production. Furthermore, the participation of developing-world fisheries in MSC certification is currently limited by the costs of entering FIP or MSC processes and of monitoring compliance with standards (Ponte 2012; Micheli et al. 2014; Blackmore et al. 2015; Sampson et al. 2015). There is already pressure on companies throughout the seafood supply chain to source MSC-certified products (Chaplin-Kramer et al. 2015). PES initiatives thus have the potential to complement and strengthen MSC certification schemes by channelling investment from these companies to help cover some of the costs to potential providers of participating in the scheme (Short 2011, 2012, 2014).

Table 2.1: Examples of marine and coastal fishery payment mechanisms documented in the peer-reviewed literature. All are based on positive incentives, but the extent of conditionality and additionality are unclear in all but the case of the European Union-Mauritania fisheries agreement.

Scheme	Start	Buyer	Provider	Payment	Ecosystem Service
Defeso: compensation for fisheries closure during fish reproduction (Begossi et al. 2011a).	1986	Brazilian Government	Artisanal fishers, Brazil	Financial compensation	Fish production
Marine Stewardship Council scheme (Roheim et al. 2011)	1997	Consumer	Certified fishery	Price premium	Meeting MSC standards
Sea turtle bycatch release (Ferraro & Gjertsen 2009).	1998	Watamu Turtle Watch: Kenyan NGO	Artisanal fishers, Kenya	Financial performance payments	Sea turtles
Sea turtle bycatch release (Ferraro & Gjertsen 2009).	2000	RENATURA: Congolese NGO	Artisanal fishers, Congo	Materials to fix/replace net	Sea turtles
Biodiversity offsets or bycatch mitigation scheme (Janisse et al. 2010).	2004	FISH: an association of California drift gillnet swordfishers	ASUPMATOMA Mexican NGO	Financial payments	Sea turtles
Vaquita bycatch reduction (Gjertsen & Niesten 2010).	2007	Mexican Government	Artisanal fishers, Northern Gulf of California	Gillnet permits purchased or leased from fishers	Vaquita population
Compensation for hilsa conservation in Bangladesh (Islam et al. 2016).	2005	Government of Bangladesh	Affected Bangladeshi communities	Compensation in the form of rice and alternative livelihood support	Fish production
International payment operating through a bilateral fisheries agreement (Binet et al. 2013).	2006	European Union	Banc d'Arguin National Park, Mauritania	Funding for the direct conservation of marine and coastal biodiversity	Biomass productivity, including protection of nursery and breeding sites for commercial species
'Reverse fishing license' programme for lost commercial fishing revenue (Lau 2012).	2008	Multiple public and private bodies acting through the PIPA* Trust	Government of Kiribati	Financial compensation	Protection of tuna spawning areas, seamounts and reefs

* Phoenix Islands Protected Area, Kiribati

2.4 Preconditions for PES in fisheries

From the terrestrial and marine literature (e.g. Wunder 2005; Sommerville et al. 2009; Tacconi 2012; Lau 2012), six key preconditions have emerged that are likely to enable the successful delivery of a fisheries PES scheme, as defined by the principles above. Ideally, there should be: 1) demand for one or more ES where supply is threatened; 2) suitable baseline data available and a set of potential management actions underpinned by robust science; 3) clarity and security of property rights; 4) capacity for hybrid multi-level governance; 5) capacity for rigorous MCS; and 6) potential for financial sustainability of the scheme.

2.4.1 Demand for one or more ES where supply is threatened

In order for a PES to deliver additional benefits, there should be some level of current or future threat to ES supply, be it overfishing or a more indirect threat. Terrestrial experience indicates that PES is most feasible when this threat is intermediate or robustly predicted (Wunder 2005). Finding buyers is likely to be more difficult in low threat scenarios where conservation benefits are unlikely to be additional, but in very high threat scenarios the opportunity costs of ES provision to providers also tend to be very high – for example, when fishers depend on a rapidly declining resource for their food security.

The more demand there is for the service(s) in question, the easier it will be to identify buyers and the greater the payment is likely to be. For example, sea turtle conservation and carbon sequestration would probably generate greater demand than demersal invertebrate biodiversity or the biological regulation provided by coral reef shark populations. Although economic valuation is by no means a prerequisite for PES, it can, in addition to giving an indication of scheme viability and appropriate design, help to promote this demand (Emerton 2013; Wunder 2013). Because of the existing market for seafood there is likely to be sufficient demand for seafood provision, particularly if the species are commercially valuable; it may be more difficult to convince potential buyers of the benefits provided by other non-market services impacted by fisheries. An investigation into the dependence of seafood companies on ES, using standardised metrics and indicators (Houdet et al. 2012), and advances in valuation methods for non-market services (Barbier et al. 2011) may help to generate demand by demonstrating the interrelatedness of seafood supply and wider ES (Chaplin-Kramer et al. 2015). Moreover, bundling closely coupled services which are in high demand and have established PES markets (e.g. carbon sequestration) with seafood production may attract buyers from outside the seafood sector.

2.4.2 Availability of suitable baseline data and robust science

For good design, implementation and monitoring, PES requires a clear understanding of the social-ecological system, and therefore the availability of baseline data on the ES in question, system dynamics, and current/previous management approaches (Wunder 2005). Although marine ES data are often lacking (Guerry et al. 2011), novel approaches are now emerging for their rigorous assessment, quantification and mapping, particularly for commercial fisheries, but also for non-market services (Chan & Ruckelshaus 2010; Tallis et al. 2012; Liquete et al. 2013; Blasiak et al. 2014). In the developing world, however, fisheries are rarely well characterised and new approaches to stock assessment, ecosystem modelling, and the incorporation of uncertainty will be required (Costello et al. 2012). For instance, low capacity for data collection might be addressed through the application of existing mobile phone technology; mobile application surveys are being trialled in the Solomon Islands with a view to improving the management decisions of its inshore fisheries (K. Rhodes, T. Welch, R. Pomeroy, M. Knight, S. Diffey and K. Simeon, unpublished data). In many developing countries even the poorest of fishers have access to mobile phones, and fishers could be compensated for using their phones to participate in data collection. When low quality or few data are available, Models of Intermediate Complexity for Ecosystem assessments (MICE) can be used to support fisheries decision-making in an ecosystem context, while accounting for a broad range of uncertainties (Plagányi et al. 2012a, b, c). They take into account ecosystem objectives, but include only the components essential to answering the management question at hand. Furthermore, PES may facilitate investment in building capacity for data collection and identification of the minimal data requirements. Alternatively, ecological data and model frameworks from well understood fisheries can be transferred and applied to data deficient situations (Chapter 5).

A clear and preferably causal link should also be established between the management action that is being paid for and service delivery, allowing for strong conditionality on outcomes (Binet et al. 2013). Yet causal relationships are dynamic and difficult to establish in marine systems (Guerry et al. 2011). Furthermore, the fluidity of the marine environment means that resources can travel quickly between management areas – particularly in the case of migratory species and those with a long-distance larval dispersal phase – making it difficult to directly link conservation outcomes with activities undertaken by the provider. Nevertheless, as discussed in detail by Lau (2012), the types of management activities which lead to enhanced service delivery are well understood in fisheries (e.g. no-take zones, seasonal closures, bycatch reduction devices) – though causal links may not always be established specifically for the system in question.

2.4.3 Clarity and security of property rights

Rights-based management systems such as Individual Transferable Quotas (ITQs) and Territorial Use Rights (TURFs) can empower fishers to use resources more sustainably and help to build institutional capacity that could benefit PES schemes, if these management systems act on the same scale as the PES (Costello et al. 2008; Kemkes et al. 2010). TURFs are likely to be more appropriate than ITQs in developing-world, coastal fisheries (Costello et al. 2012).

However, in small-scale and artisanal fisheries, where recognition and enforcement of rights often fails, PES has the potential to facilitate the process of rights clarification, which can in turn act as a precursor to other forms of management (Clements et al. 2010; van Noordwijk et al. 2012). For example, Janssen et al. (2007) have proposed that PES might be used to address rights allocation issues and achieve both fishery sustainability and protection of traditional fishing communities in South Africa by compensating artisanal fishers for limiting their catch with revenues from commercial fishery license fees. Success would depend, however, on the interdependencies of those two fisheries and the sustainability of the commercial fishery. PES schemes might also be established under open-access or common property regimes, as they have been in terrestrial systems, by providing incentives for collective action and formalising customary rights (Greiber 2009; Clements et al. 2010; Muradian 2013b; Midler et al. 2015). Moreover, in Lower Amazon floodplain fisheries, the creation of co-management Fishing Agreements (FAs) has defined some access rules and established a legal institutional framework, a process which could in theory be applied to provide an institutional basis for PES in other coastal fisheries (Hallwass et al. 2013). On the high seas, which are essentially devoid of rights, PES is not currently thought to be feasible (Lau 2012). However, the concept of 'side payments' is being explored for internationally shared tuna stocks (Bailey 2013), and with the right cooperative institutional arrangements and industry-wide agreements, a high seas PES scheme might help address current weaknesses in high seas governance (Aqorau 2007; Blasiak et al. 2014; Visbeck et al. 2014).

2.4.4 Capacity for hybrid multi-level governance

A PES mechanism requires a good governance structure if it is to be sustainable, provide the accountability mechanisms to facilitate payments to the correct providers, increase transparency, and reduce transaction costs (Chapter 7; Vatn 2010; Wunder 2013). Due to the highly mobile and dispersed nature of most marine resources and stakeholders, hierarchical (e.g. state) and market governance structures will rarely be suitable in marine PES schemes (Muradian & Gómez-Baggethun 2013). Muradian (2013a) argues that hybrid forms of governance such as collective management, which sit between markets and hierarchies, will be most efficient in marine systems, although this will depend on the institutional context (Clements et al. 2010; Vatn 2010; Ingram et al. 2014).

Although PES is often viewed as an alternative to the failure or lack of conventional regulatory measures, government regulation can be complementary and synergistic (Habtezion 2013), and PES is often led by governments (Engel et al. 2008). Whether it is a public scheme or not, there must be support of the PES from the government in the state(s) where the fishing takes place (Binet et al. 2013). The variability of fish markets and subsequent variability of opportunity costs mean that in some circumstances a fisheries PES scheme would probably fail without regulation (Barr & Mourato 2009; van Noordwijk et al. 2012). Moreover, if fishing rights are ambiguous, governments may need to assist with their clarification and enforcement. Where national governance is poor, however, local champions of fisheries management, collective institutions, and systems of community governance could increase the opportunity for and durability of a PES scheme (Clements et al. 2010; Gutiérrez et al. 2011; Begossi et al. 2012; Carbonetti et al. 2014; Ingram et al. 2014). These institutions may in turn benefit from the introduction of PES if collective payments included, for instance, funding of training to improve internal collaboration (Hallwass et al. 2013). The pre-existence of certain locally developed institutions like co-management schemes may thus prove an advantage for PES (Chapter 7; Vatn 2010; Kemkes et al. 2010; Ingram et al. 2014). However, the formation of new multi-level institutions may also be necessary to drive coordination between relevant stakeholders at multiple scales (van Noordwijk et al. 2012; Habtezion 2013). At a regional or national scale, this might entail the integration of community-level organisations, government agencies, NGOs and the private sector (Bailey et al. 2015). On an international scale, a PES might be integrated into pre-existing multilateral environmental agreements like the UN Convention on the Law of the Sea (UNCLOS), which would benefit from increased international coordination (Habtezion 2013; Visbeck et al. 2014). Lessons in multi-scale PES governance can be drawn from initiatives to reduce emissions from deforestation and forest degradation (REDD+; van Noordwijk et al. 2012).

2.4.5 Capacity for monitoring, control and surveillance

With good governance there should come rigorous systems of MCS. Whether a PES scheme is outcome- or action-based, the technical and financial challenges of MCS are heightened in the marine environment (Lau 2012). The requirement of conditionality under PES may raise transaction costs compared to other management approaches. However, MCS is vital for sustainable fisheries management, regardless of whether PES is in place, and many management agencies already have the equipment and resources required. In low-income developing countries where this is not the case, there are alternatives that would reduce transaction costs. For example, if a scheme were established under the common property rights of a fishing community, conditionality could be achieved through social norms which emerge from collective action (Clements et al. 2010), or through active community-level enforcement (Begossi et al. 2012).

If payments are conditional, they should therefore incentivise investment in MCS so that providers qualify for these payments. Furthermore, PES could be designed explicitly to alleviate or cover the costs of MCS. For example, collective providers of PES could use their payments to cover the cost of boats and fuel or for the training or participation of community members (Hallwass et al. 2013). It has been proposed that coastal artisanal fishers participating in compensatory mechanisms could use their local knowledge to monitor fishing sites used by industrial fishers in order to reduce the catch of juveniles and supplement landings data

(Begossi et al. 2011b). On an international scale, through the EU-Mauritania Fisheries Agreement and PES scheme, the EU is investing in MCS of the Banc d'Arguin National Park (Binet et al. 2013).

2.4.6 Potential for financial sustainability

When PES connects direct beneficiaries of ES with ES providers through a closed loop system in which environmental externalities are internalised, it should in theory be financially sustainable (Engel et al. 2008; Farley & Costanza 2010). Governments, however, often act on behalf of beneficiaries, in which case financial sustainability may depend solely on budget allocations (Wunder et al. 2008). Furthermore, when payments are for biodiversity conservation, which is inherently more abstract than provisioning and supporting services (Morar et al. 2015), the loop between buyers and providers is less tangible, and ongoing payments may be less secure. In these circumstances, there must be a mechanism in place for financial sustainability – whether this is established through a tool which generates a constant flow of finances, for example in the form of user fees or taxes on fishing license fees, or one which generates revenue from investments in ES provision (e.g. impact investment, see Chapter 7). It is likely that any investment in an ES related to seafood provision will increase fishery profitability, whether this is a result of stock recovery or the ability to set a price premium on an end product. Yet this may not be the case if, for instance, the action being paid for is one which may not affect fishery profits (e.g. bycatch reduction). The development of a Conservation Trust Fund (CTF) may enhance the financial sustainability of PES in such cases (Chapter 7; Goldman-Benner et al. 2012). A CTF, where payments are made using the revenue from investment or a portion of principal funds, can act as an intermediary between buyers and providers and thus enable the use of tools such as endowments and revolving funds to create a sustainable source of financing for payments (Bladon et al. 2014a). Though the majority of experience comes from terrestrial conservation, trust funds are being used to generate and allocate funds in marine conservation agreements, MPA management plans and some marine PES schemes. For example, the creation of a CTF for Mauritania's marine PES scheme has played a role in attracting investment (Binet et al. 2013).

2.5 An exploration of the applicability of PES in real world circumstances

In order to assess whether PES could actually satisfy its defining principles and be successfully implemented in a real developing-world fisheries management situation, I explore the presence or absence of the above preconditions in four contrasting case studies (Table 2.2). In each case, the potential configurations and contributions of PES are discussed and the suitability of and barriers to such an approach are analysed, thereby highlighting the strengths and weaknesses of PES under a broad range of institutional and ecological circumstances. The case studies are: a relatively simple and well-managed domestic whitefish fishery; a complex multi-sector domestic shrimp fishery; a lucrative international fishery targeting a highly migratory species; and a multi-sector and poorly understood domestic fishery targeting a largely anadromous fish.

Table 2.2: Potential structure of PES mechanisms in three contrasting fisheries and a summary of the extent to which they fulfil the design preconditions for marine PES. '✓' indicates complete fulfilment of a precondition; '✓×' indicates partial fulfilment; '×' indicates failure.

	Namibian hake fishery	**Mozambican shallow water shrimp fishery**	**PNA skipjack tuna fishery**	**Bangladesh hilsa fishery**
Potential PES configuration				
ES / management action	Hake production / bycatch reduction	Mangrove or seagrass habitat / shrimp production.	Skipjack production / reduction of juvenile & non-tuna bycatch.	Fish production (hilsa & other food products) / range of regulatory & supporting services of coastal habitat.
Buyers	International seafood companies; European retailers; Governments of Namibia, South Africa & Angola; international consumers.	International seafood companies; European retailers; commercial fishing companies; international consumers; tourism, oil & gas industries.	PNA, tuna retailers; Pacifical; seafood companies; international consumers; longline fleets.	Commercial marine fishers; Governments of Bangladesh, India & Myanmar; international consumers; national & international companies.
Providers	Namibian and South African hake fleets.	Artisanal fishing communities in Mozambique.	PNA skipjack fleet.	Artisanal river fishing communities in Bangladesh.
Design preconditions				
Demand for & threat to ES	✓×	✓	✓	✓
Baseline data & science	✓×	×	✓	×
Property rights	✓	✓×	✓	×
Capacity for governance	✓	×	✓×	✓×
Monitoring & enforcement	✓	✓×	✓	×
Financial sustainability	✓×	✓×	✓	✓×

2.5.1 Namibian hake fishery

This industrial demersal trawl and longline fishery, targeting cape hake (*Merlucius capenis*) and deep-water hake (*Merlucius paradoxus*), is the country's most valuable fishery and almost entirely exported (OECD 2012). Although it is widely thought to be relatively well managed, stocks appear to be in decline and issues of social equity have been highlighted (Paterson et al. 2013). A PES scheme might improve management by realigning social, ecological, and economic goals.

Demand for one or more ES where supply is threatened

There is evidence of a decline in hake stocks in Namibia (BCC 2011) and global demand is high, so a short-term transitional and international PES scheme may be appropriate. This could be expanded to pay for bycatch reduction, but whitefish fisheries tend to deliver relatively few other ES and opportunities for bundling may therefore be limited. In turn, this reduces the potential for PES to produce additional ecosystem benefits over conventional management approaches in the long term.

Availability of suitable baseline data and robust science

After decades of overexploitation, Namibia has rebuilt its hake fishery through access limitation and a reduction in effort, and the impacts of these actions are well established. Reliable stock assessments, strong ecosystem understanding, and modelling approaches are available, which could be used to support conditionality on ecosystem outcomes (Roux & Shannon 2010; Kirchner et al. 2012). However, more detailed and reliable social data might be required to ensure the equitable distribution of benefits (Paterson et al. 2013).

Clarity and security of property rights

Namibia has developed a clear rights-based legal framework and distributed individual non-transferable quotas (Armstrong et al. 2004). This policy aims to ensure that social benefits are returned to citizens, but has been criticised for favouring the elite minority over local development (Paterson et al. 2013) – an issue which any PES scheme should address. Nevertheless, the pre-existence of clear and secure access rights would make the identification of buyers and providers relatively simple, considering that the marine fisheries sector is exclusively industrial and conflicts with other users are unlikely.

Capacity for hybrid multi-level governance

A PES approach would fit within the perspective of current fisheries management. Despite being a developing country with limited financial resources, Namibia has made a formal commitment to ecosystem-based fisheries management (Roux & Shannon 2010; MFMR 2012) and the Government has already shown interest in a participatory and incentive-based approach, as well as in maximising the value returned to Namibians (the allocation of fishing rights was itself driven by tax reductions connected to quota fees). Although they hardly enter international waters, both species of hake are managed together as a single stock which is shared with South Africa and Angola, so a PES scheme would require international cooperative governance – a prospect which is currently being put into action through the Benguela Current Convention and Commission (MFMR 2012).

Capacity for MCS

With the majority of fishing rights owned by Namibians, users are proximate and not too high in number; and there is a good MCS system already in place (Bergh & Davies 2004; OECD 2012). This should reduce transaction costs and allow benefits to be directly linked back to the activity which is being paid for.

Potential for financial sustainability

Buyers could be promised a return on investment through increased fishery profitability. Furthermore, the Namibian Government currently collects catch levies for a 'fisheries fund' which is used to finance fisheries research (OECD 2012), although it is unclear whether it would have the institutional capacity to administer a PES.

Verdict

The hake fishery satisfies most of the preconditions: it has clear and secure property rights, the beginnings of a hybrid multi-level governance system, good MCS and a potential mechanism for financial sustainability. Nevertheless, there may be limited opportunities for long-term additionality of a PES scheme.

2.5.2 Mozambican shallow water shrimp fishery

Mozambique's shallow water shrimp fishery (Indian White prawn *Fenneropenaeus indicus* and Speckled shrimp *Metapenaeus monoceros*) is a commercially important and multi-sector fishery (Palha de Sousa et al. 2011). It fulfils some of the preconditions for a PES scheme, which has the potential to address the need for a holistic management approach, but there are a number of social and ecological uncertainties which may limit implementation.

Demand for one or more ES where supply is threatened

Although the fishery is considered to be relatively sustainable, there is overcapacity, conflict between the artisanal and commercial sectors, and high levels of non-shrimp bycatch and discarding (Banks & Macfadyen 2011). For example, continued expansion of the artisanal sector, in which there is juvenile overfishing, could have serious economic implications for the semi-industrial sector (Palha de Sousa et al. 2011). If a PES scheme were to operate through payments from downstream supply chain companies to artisanal fishers – for instance, as compensation for avoiding nursery grounds in a marine reserve – inter-sector conflict might be reduced. Economically, the shrimp fisheries are the most important in Mozambique, and the artisanal sector is the principal source of food and employment for coastal communities (Banks & Macfadyen 2011). Local and foreign demand for shrimp from producers, retailers, and consumers is therefore high, and buyers could be convinced of the importance of mangrove and seagrass habitats in this production. The potential for mangrove-based PES systems is well recognised (Locatelli et al. 2014), and if services were bundled it might also be possible to attract investment from other industries such as tourism or oil and gas, and thus the global carbon market.

Availability of suitable baseline data and robust science

This ecosystem is not well understood, and fishery success in any one year is likely to be strongly affected by variation in freshwater flow from the Zambezi River and its effect on recruitment (Gammelsrod 1992), making a direct link between a particular management activity and ecological outcomes difficult to establish. Artisanal catches are not yet incorporated into annual stock assessments, although they are estimated to form a substantial proportion of the total shrimp catch, and data on the number of artisanal vessels in operation are highly inconsistent (Banks & Macfadyen 2011). The resulting uncertainty in catch data is one of the main reasons why a pre-assessment for MSC certification found the fishery to be unsuitable in its recent state (Moody Marine Ltd 2008). However, the requirement for conditionality in PES may incentivise investment in innovative new methods of artisanal data collection. It might also be possible to use models developed for other fisheries as a framework for the Mozambican shrimp, given that there has been extensive work conducted on the well managed and well understood multi-species Australian northern prawn fishery, which incorporates the issue of uncertainty (Dichmont et al. 2006a, 2006b, 2006c).

Clarity and security of property rights

The fishery is contained within Mozambique's Exclusive Economic Zone (EEZ), and commercial access is restricted to licence holders. Furthermore, the Government is moving towards a rights-based approach, which would make the system more amenable to PES (Tindall 2012). However, there is a great deal of interaction between the shallow and deep water fisheries in the region, and a PES scheme should probably not be limited to the shallow water fishery. The artisanal sector is essentially open access and supports several thousand fishers along the coastline (Banks & Macfadyen 2011), so making the identification of providers complex. Nevertheless, local co-management committees operating licensing systems have now been established in most fishing communities, with some success, which might provide an institutional basis for PES (MRAG 2010).

Capacity for hybrid multi-level governance

Through a comprehensive management plan, the Government is beginning to address management concerns, but it still lacks implementation capacity. Furthermore, the interactions between the shallow and deep water fisheries in the region – and between the commercial and artisanal sectors of the shallow water fishery – are not currently reflected in their governance. A PES scheme would require the development of institutional capacity through the coordination and integration of management authorities in different sectors, regions and at different levels, as well as with corporations and co-management organisations (Banks & Macfadyen 2011).

Capacity for MCS

For a basic level of conditionality to be achieved, the current system of MCS would need improvement (Banks & Macfadyen 2011). Illegal, unreported and unregulated (IUU) fishing is a problem in Mozambique (MRAG 2005), the industrial Vessel Monitoring System (VMS) is not fully operational (Banks & Macfadyen 2011) and the artisanal fishery is largely unregulated. Artisanal fishers are not included in the seasonal closures imposed on commercial sectors, and they operate close to shore, often in estuarine mangrove nursery areas, and increasingly using illegal mosquito nets (M. Rodrigues 2013, WWF Mozambique, personal communication, 22nd April.) Low levels of national educational capacity and access to technology in Mozambique may mean that considerable investment in capacity building would be required to achieve the necessary improvements in top-down research, administration and enforcement (FAO 2007). A system of co-management may be preferable (Béné et al. 2010) and it could be possible for a PES scheme to be designed so as to include mechanisms for alleviating the social costs of community monitoring and enforcement (Begossi et al. 2011a).

Potential for financial sustainability

Buyers could be promised a return on investment through increased fishery profitability and there may be opportunities for mechanisms which generate a constant flow of revenue through user fees.

Verdict

Application of PES to the shrimp fishery in its current state may in theory be limited by poor ecosystem understanding and a lack of institutional capacity for integrated governance, MCS, and financial sustainability. But through investment in capacity building and the identification of gaps in understanding, PES might provide a mechanism to address these deficiencies.

2.5.3 PNA Western and Central Pacific purse seine skipjack tuna fishery

The application of PES to tuna fisheries would be ideal in terms of their lucrative markets, but also particularly challenging in terms of the highly migratory nature of tuna. The PNA (Parties to the Nauru Agreement) is a sub-regional alliance between eight adjacent Pacific island states for the management of a purse seine skipjack tuna (*Katsuwonus pelamis)* fishery. Although the institutional and political complexities of the Western and Central Pacific Ocean (WCPO) tuna fisheries might on first glance render them ineligible for PES, the PNA has created an environment in which PES could be feasible.

Demand for one or more ES where supply is threatened

Tuna has high economic value (Collette et al. 2011) and demand is therefore great from consumers, international retailers, producers and the economies of dependent Pacific states. PNA skipjack is considered to be sustainably managed and the free-school skipjack fishery has gained MSC certification (Moody Marine Ltd 2011). However, those fleets that still target skipjack near Fish Aggregating Devices (FADs) catch undersize tuna and contribute to the overfishing of other species such as bigeye tuna (*Thunnus obesus;* Bailey 2013). A PES targeting skipjack production and/or bycatch reduction might generate additional benefits over those delivered by current PNA management. Payments to PNA skipjack fleets channelled from various potential sources, including longline fleets which target bigeye, could incentivise free school purse seining.

Availability of suitable baseline data and robust science

Good stock assessments and data on skipjack and bigeye movements are available, and relatively advanced spatial ecosystem and population dynamics models can also be used for the investigation of tuna management scenarios and inter-species interactions in the Pacific (Lehodey et al. 2008).

Clarity and security of property rights

WCPO tuna fisheries fall under the jurisdictions of multiple states which hold property rights within their EEZs, and fishing operations range from coastal artisanal fleets to industrial purse seine and longline fleets on the high seas (Aqorau 2007). The variable movements of tuna therefore complicate the identification of buyers and providers and may create a free-rider problem: one buyer might end up paying for conservation measures while a distant actor reaps the benefits (Bailey 2013). However, the PNA states have asserted their property rights to control 70 per cent of WCPO and 50 per cent of global skipjack tuna catch, and established high seas closures in the areas between their EEZs (Maurice Brownjohn 2013, Commercial Director of PNA, personal communication, 28th April.) The PNA also operates a Vessel Day Scheme (VDS) for purse seiners. This is a rights-based transferable effort scheme whereby PNA members are allocated fishing days which they can trade among themselves and allocate to distant water fleets at their discretion (Aqorau 2007; Havice 2013). The strengthening of rights which this entails would make it possible to identify providers clearly in a PES regime.

Capacity for hybrid multi-level governance

PNA was established following the recognition of mutual concerns among the participating states over sustainability and economic opportunity; and there has since been a paradigm shift in the way they view and manage their resources. In addition to political will of individual states, the PNA forms an institutional basis for the implementation of a PES scheme, although the complexity of creating such a large-scale and international PES cannot be underestimated. PNA resources are not located solely within the PNA states, but fall under the jurisdictions of multiple and overlapping regional management and advisory bodies, including the WCPFC (Western and Central Pacific Fisheries Commission), FFA (Pacific Islands Forum Fisheries Agency), PNA, and TVM (Te Vaka Moana). A multi-level governance framework for the integration of each of these bodies would be desirable, but mutual economic and ecological objectives would first need to be identified.

Capacity for MCS

PNA terms of license enforce strict management rules with 100 per cent observer coverage (M. Brownjohn 2013, Commercial Director of PNA, personal communication, 28th April) and the VDS provides a mandatory real-time Vessel Monitoring System (Havice 2013), which might allow payments to be conditional upon actions or outcomes. Enforcement is difficult on the high seas, but PNA has also established high seas closures in areas between EEZs, reducing IUU and, therefore, the risk of free-riding.

Potential for financial sustainability

PNA management is funded through fixed conservation levies from fishing vessels; and although these levies are not currently conditional on meeting any specific ecosystem objectives or MSC standards, this could provide the basis for a sustainable PES. Furthermore, there is already strong supply chain and market interest in the sustainability of the fishery. Thanks to the market access enabled by its joint venture with import and branding agency Pacifical, the PNA generates a 20 per cent premium on free school skipjack products via MSC certification and labelling, the majority of which is channelled back to the industry. Its co-branding programme with Pacifical also allows traceability of a can of tuna back to the vessels and factory involved in its production (Tindall 2012) - a system which has created an unprecedented level of transparency within the supply chain and which could allow corporate responsibility to be apportioned accordingly (Short 2012).

Verdict

This case study fully satisfies five out of six of the preconditions. The only theoretical barrier to PES is the complexity of WCPO tuna governance; increased levels of coordination between PNA and other governing bodies may not be a realistic expectation.

2.5.4 Bangladesh hilsa fishery

Hilsa (*Tenualosa ilisha*) are largely anadromous, the majority of the population migrating from marine to fresh water for spawning, and as such support riverine and marine fisheries in Bangladesh (Blaber et al. 2003b; Amin et al. 2008). The Bangladesh Government currently runs a compensation scheme for hilsa fishers, which is designed to improve their socioeconomic positions in the face of fishing regulations; but there are questions surrounding the effectiveness, efficiency, and equity of the scheme (Chapters 5 & 6; Islam et al. 2016), which might be addressed through a more formalised PES approach (Mohammed & Wahab 2013).

Demand for one or more ES where supply is threatened

The hilsa fishery is the largest and most valuable single-species fishery in Bangladesh, but there has been a decline in stocks in recent years (Mome & Arnason 2007). As a fish with historical significance in Bengali culture, there is strong demand for hilsa, both locally and internationally amongst the Bengali diaspora (Chapter 3). As such it supports the livelihoods of up to 500,000 fishermen and an additional 2-2.5 million workers involved in the supply chain (Haldar 2004; Mohammed & Wahab 2013; Islam et al. 2014). Although there may be less demand for hilsa from international companies than for seafood which is more popular in the west, the Government has already demonstrated a willingness to invest in sustainability improvements through its current management approach, which recognises the socioeconomic cost imposed upon hilsa fishers during fishing bans. The bundling of other ES into a PES scheme would increase investment opportunity outside of the seafood sector. For example, small-meshed gillnets used to catch hilsa are associated with aquatic biodiversity issues in the riverine fishery (BOBLME 2012), and the coastal habitats protected within the hilsa sanctuaries provide regulating and supporting services, including those that contribute to the resilience of fisheries and coastal communities to climate change (Allison et al. 2009). There is potential for payments for hilsa conservation to be integrated with other social and environmental protection schemes (Chapter 8).

Availability of suitable baseline data and robust science

Hilsa are poorly understood; stock assessments in Bangladesh are highly unreliable; and there is little baseline social, economic, or ecological data available, particularly for the riverine fishery (Chapter 3; Mome & Arnason 2007; Amin et al. 2008; BOBLME 2010). Hilsa are threatened not only by overfishing, but by pollution, upstream damming, and climate change; and yet research on the impacts of non-fishing threats is limited (Blaber et al. 2003a; BOBLME 2010). Reported declines in production prompted the Government to designate sanctuaries in major riverine and coastal nursery grounds, in which seasonal fishing bans are implemented for the protection of juveniles (Chapter 3). However, the scientific basis for this approach is weak, and it has not been possible to link causally any changes to the compensation scheme or associated management (Chapters 4 & 5). On the basis of data currently available, ecological additionality may be difficult to monitor directly, and payments may need to be conditional on actions rather than outcomes (Chapter 5). Even so, a PES could be undermined by the rapid environmental changes that are affecting the hilsa fishery.

Clarity and security of property rights

The process of buyer/provider identification would be complex due to the migratory nature of hilsa and the open access nature of the fishery. There is a lack of recognition of traditional property rights and most vessels are unregistered (Chapter 3; Amin et al. 2008; BOBLME 2010). Identification cards are being issued to help the authorities distinguish genuine fishers from those claiming to be so within Bangladesh (Chapter 5), but hilsa stocks in the Bay of Bengal are also commercially exploited by Myanmar and India (BOBLME 2010). Not only would these countries benefit from measures implemented by Bangladesh, but their river fishery management will in any case affect stocks in Bangladesh. The prospects of transboundary management is being discussed through the Bay of Bengal Large Marine Ecosystem Project (BOBLME 2012), and a cooperative transnational PES system would therefore be desirable, though perhaps not politically feasible at present (Chapter 7).

Capacity for hybrid multi-level governance

Current management in Bangladesh indicates that there is political will for the governance of a national PES scheme. The Government has taken up management recommendations made by a number of external projects (BOBLME 2010), and currently makes budget allocations to the compensation scheme itself (Mohammed & Wahab 2013). Linkages between the various agencies involved in hilsa management are, however, weak (BOBLME 2012), and PES would require more cooperation between institutions at multiple levels (Chapter 7). Risks posed by potential political instability should also be considered, though these might be mitigated by decentralisation of management (Chapter 7). Ideally, a multi-level institution should be established among Bangladesh, India and Myanmar, integrating the relevant national and local governments and agencies, NGOs, and community level organisations.

Capacity for MCS

Compliance with conservation measures in the hilsa fishery is low and MCS is poor (Chapter 5; Siddique 2009; BOBLME 2010). Although conditionality may therefore be difficult to achieve, the implementation of PES could help to build capacity for MCS. The scheme might be designed to involve local communities and fisher associations in monitoring and enforcement; collective payments might be used to invest in mobile phone technologies to provide more reliable data from the artisanal fishery; and compliance might be monitored indirectly through surveys of fisher perceptions over time (Hallwass et al. 2013).

Potential for financial sustainability

Although compensation currently comes directly from Government budget allocations, the creation of a CTF for hilsa conservation is, at the time of writing, underway, which could generate new and more sustainable sources of finance for a PES (Chapter 7; Mohammed & Wahab 2013).

Verdict

The hilsa fishery appears to be the least amenable case study, satisfying only one of the six preconditions fully. However, in these less-than-perfect circumstances PES might be used to leverage the changes required. The recent political support for a sustainable increase in hilsa production indicates great potential for a national PES scheme to facilitate investment in capacity building for community-based monitoring and enforcement, as well as the development of a more cooperative and integrated system of governance.

2.6 Conclusions

There are challenges to the design and implementation of PES in marine environments, but too much focus on theory may restrict and obscure opportunities for unconventional PES schemes to benefit fisheries management and stakeholders (Muradian et al. 2010; Farley & Costanza 2010). PES is not a silver bullet, but when used together with conventional regulatory approaches it could play a significant role in incentivising sustainable fishing practices in developing countries. Furthermore, through systematic private sector engagement, PES could facilitate increased investment in fisheries improvement by buyers of ES, thereby reducing the burden of costs and responsibility placed on governments and fishers themselves.

First and foremost, payments should be conditional on actions, if not on conservation outcomes, but these actions still need a robust scientific basis. Six key preconditions have been identified which would, in theory, enable the successful delivery of a fisheries PES scheme. It is rare that a developing-country fishery will fulfil each of these preconditions *a priori*, but that is not to say a

PES approach would be inappropriate in such circumstances, as long as there is capacity for improvement (Micheli et al. 2014). In practice, design will depend on the precise institutional context and may require creative and innovative approaches to the maintenance of conditionality and additionality. For example, the Bangladesh hilsa fishery satisfies few PES preconditions, but the level of political will already displayed for sustainable management presents an opportunity for improvement through the support of a PES (see Chapter 7). Although the hilsa fishery lacks a history of rights-based and ecosystem-based management as demonstrated by the Namibian hake and PNA tuna examples, it does have a compensation scheme already in place, and any conjunction between existing institutions and those required for PES increases the viability and sustainability of such a scheme. For highly migratory species like tuna, the development of international collaborative institutions such as the PNA may play an important role in the feasibility of PES, though they must grow in scale if they are to operate on the high seas. Where weak governance and institutions reduce the potential for conditionality, as is often the case in developing countries, PES could drive institutional reform; it might, for instance, be used to facilitate rights clarification, to pay for or incentivise improvements in MCS, or to address the shortcomings of an existing co-management programme. More research at the interface of PES and complementary institutional structures such as CTFs, TURFs, ITQs and co-management systems could encourage and facilitate the appropriate application of PES in developing-world fisheries.

Chapter 3

Creating a frame of reference for hilsa management and conservation interventions in Bangladesh

3.1 Introduction

An effective conservation intervention should have a measurable conservation benefit, and thus requires the specification of an appropriate frame of reference against which it can be evaluated (Maron et al. 2013; Bull et al. 2014). A frame of reference is an umbrella term that may refer to fixed baselines (e.g. current conditions or a past reference state), dynamic baselines (dynamic scenarios reflecting background rates of change) and counterfactuals (projected scenarios estimating what would have occurred in the absence of an intervention; Bull et al. 2016). Truly rigorous evaluations of conservation impact require the development of a frame of reference that includes not only a baseline understanding of the current status of the conservation target, but also the projection of a counterfactual (Ferraro & Pattanayak 2006; Bull 2014; Bull et al. 2015). It is the counterfactual that enables the measurement and attribution of true conservation impact; i.e., the difference between the outcome of the intervention and the estimated outcome in the absence of the intervention.

The frame of reference should include at least two dimensions of environmental change: ongoing trends in the ecological status of the conservation target, and anthropogenic impacts upon this target (Bull et al. 2014, 2015). The development and evaluation of any conservation intervention should also consider the social-ecological and historical contexts in which it operates – a perspective that has been slow to reach the fisheries management community (Ostrom 2007; Pooley 2013; Hilborn et al. 2015). Ideally, therefore, a frame of reference should include ecological trends; institutional, social, economic and physical factors; and the potential interactions and feedbacks between them (Ferraro & Pattanayak 2006; Nicholson et al. 2009).

Impact evaluations using experimental and quasi-experimental methods are emerging in the conservation literature (Andam et al. 2008; Pattanayak et al. 2010; Clements & Milner-Gulland 2014). However, few examples exist of counterfactuals being developed early in the intervention design process, and when they are, they are often developed with incorrect assumptions or assumptions that are not made explicit (Maron et al. 2013). A common reason for this is lack of data; counterfactuals are subject to numerous sources of uncertainty and it can be challenging to develop and validate projected trends when knowledge is poor (TEEB 2010; Bull et al. 2014, 2015).

In fisheries management, uncertainties limit the ability of policymakers to project trends and to predict the effects of management interventions (Davies 2015). These challenges are most pronounced in small-scale and developing world fisheries, where data limitations mean even fixed baselines can be difficult to estimate (Carruthers et al. 2014). Nevertheless, useful counterfactuals can be developed even in these circumstances, as long as assumptions and limitations are acknowledged (Bull 2014; Bull et al. 2015). The process of developing baselines and counterfactuals can actually thereby highlight key areas of uncertainty which might hinder the development of effective interventions and our ability to evaluate them.

In this chapter I demonstrate the value of this process by developing a frame of reference for the hilsa (*Tenualosa ilisha*) fishery in Bangladesh, which could be used to evaluate current or potential hilsa management and conservation interventions. I combine qualitative and some quantitative analyses of secondary datasets and literature in a qualitative way to explore a) patterns of social, economic, institutional and physical change relevant to the management of hilsa in Bangladesh; and b) ecological trends in the hilsa fishery.

3.2 Methods

I followed the framework of Bull et al. (2015) for the development of a frame of reference. I first set the context with a brief history of Bangladesh since its independence in 1971, before exploring the potential drivers of ecological change in the hilsa fishery: institutional, social, economic and physical. I then looked for trends in hilsa abundance and distribution from the available literature and secondary data. I compiled and analysed secondary data taken from published literature, online sources and collected by Bangladesh Agricultural University. Based on expectations formed through this analysis, I developed a conceptual map of the potential interactions between these drivers and attempted to combine them into a useful frame of reference. I outlined what I expected to be key interactions between hilsa and key factors and created a set of potential counterfactuals that consist of projections based on the identified trends. I did not explore in depth the interactions between current management and hilsa trends because this is the focus of Chapter 5, which assesses the potential ecological additionality of current management.

3.2.1 Potential drivers of change

I reviewed the literature and assessed secondary datasets for trends in those factors that I hypothesised could affect hilsa abundance and distribution, and I conducted statistical analyses using linear models where appropriate.

Institutional drivers

I explored the current national institutional and legislative context of hilsa management.

Social drivers

Human population data for Bangladesh were obtained online and poverty trends were taken from the Bangladesh Poverty Assessment (World Bank 2013; UN 2015b). I expected population size to be relevant as a factor which may influence natural resource use. I also expected livelihood and poverty trends to be relevant because low income and a lack of alternative livelihood options have been linked to illegal fishing activities (Chapter 6).

Economic drivers

I explored the economy of Bangladesh, identifying trends in sectors that would be expected to impact the hilsa fishery, whether through their role in supporting livelihoods and potential subsequent implications for fishing pressure, or through potential direct or indirect environmental impacts on hilsa populations. Because of the direct impact of exploitation on abundance I focused on bioeconomic trends in the fishing industry, looking for trends in commercial catch per unit effort (CPUE) and economic value in the hilsa fishery.

Physical drivers

Fish population variability is closely linked to environmental variability (Lehodey et al. 2006), and the life history of hilsa is known to be influenced by environmental conditions (Rahman et al. 2012a; Ahsan et al. 2014). Since climate change is therefore a potential driver of change in the hilsa fishery (Klyashtorin & Lyubushin 2007), temperature and precipitation data were obtained from five meteorological stations within the Meghna Estuary case study area used throughout this thesis (see Appendix B.1). I also considered sea-level rise, which can be linked to salinity and flooding (Ali 1999; Agrawala et al. 2003; Miah 2015); siltation and water diversion activities, which may cause morphological and hydrological changes that can be linked to reduction in availability of inland hilsa habitat (BOBLME 2010; Miao et al. 2010; Rahman et al. 2012b); pollution, which can affect the quality of freshwater and marine habitats and is considered a threat to the inland hilsa fishery (Islam 2003; Das 2009; BOBLME 2012); and deforestation, which might be linked to flooding, salinization and (if upstream) to turbidity and siltation.

3.2.2 Hilsa trends

I compiled secondary data for parameters underlying movement, reproduction, growth and mortality, and explored trends in the abundance and spatial distribution of hilsa in Bangladesh, which these parameters partially determine. I included national catch statistics (and CPUE, where available), which can be useful indicators of abundance when other data are sparse (Pauly 2013).

3.3 Results

3.3.1 Brief recent history

The People's Republic of Bangladesh gained independence from Pakistan at the end of the Liberation War in 1971 and has since experienced rapid political, socioeconomic and environmental change (Fig. 3.1). Famously dubbed by Henry Kissinger a 'basket case', following this independence Bangladesh suffered from famine, natural disaster and military rule, before the restoration of parliamentary democracy in 1991. Administration has since been passed back and forth between the Bangladesh Nationalist Party (BNP) and the Awami League, but due to widespread corruption, institutional politicisation and misallocation of resources, governance is poor (BTI 2012). This has allowed developmental NGOs – through their partnership with western aid agencies – to dominate the rural economy and act as a powerful 'shadow state' (Karim 2008). The political climate has remained tense and following the International Crimes Tribunal, set up in 2009, tensions mounted and led to the disruption of transport and movement of goods (BTI 2012). Nevertheless, Bangladesh has made remarkable social progress and become a model for other developing countries (Asadullah et al. 2014).

Numerous environmental acts and supporting policies have been introduced during the process of rapid industrialisation and urbanisation triggered by independence – many of which are relevant to the hilsa fishery (Fig. 3.1). For example, the National Environmental Policy of 1992 emphasised the conservation and management of fisheries, and the National Environmental Conservation Rules of 1997, introduced following the 1995 Act, emphasised the mitigation of industrial water pollution. The Government has taken a great deal of interest in the hilsa fishery since the 1980s, developing various policies that are discussed in detail in Section 3.3.2 (Islam et al. 2016). However, in general, legislation has not kept pace with environmental change and it is often poorly implemented (Clemett 2004; BOBLME 2011a).

Direct		Indirect
	1970	• Independence from Pakistan
• Territorial Waters and Maritime Zones Act • BFDC established under the Bangladesh Fisheries Development Corporation Act	**1975**	• Vulnerable Group Feeding programme established following 1974 famine • Environmental Pollution Control Ordinance
• Protection and Conservation of Fish Ordinance • Marine Fisheries Ordinance[4] • Fish catch assessment survey initiated	**1980**	• Army coup
• Protection and Conservation of Fish Rules, 1985[5].	**1985**	• New Fisheries Management Policy • BNP founded
• BFRI research on hilsa starts and lowest hilsa catch levels	**1990**	• Restoration of democracy by BNP • National Environmental Policy
• Fourth Fisheries Project begins	**1995**	• National Environmental Management Plan • Awami League Government • Environmental Conservation Act & Rules • National Fisheries Policy • National Water Policy and Management Plan
• Low hilsa catch levels • HFMAP implemented • *Jatka* fisher rehabilitation programme • begins	**2000**	• Coalition Government • Environmental Court Act
• Hilsa export banned	**2005**	• Coastal Zone Policy and Strategy • Awami League Government • Mobile Court Ordinance • International Crimes Tribunal set up
• BOBLME Programme begins • IIED Darwin Initiative begins • Fisher ID cards phased in • ECOFISH project begins	**2010**	• Oil spill
	2015	

Figure 3.1: Timeline of key institutional, social, economic and ecological events in the recent history of Bangladesh classified by potential direct and indirect relevance for hilsa. BFDC = Bangladesh Fisheries Development Corporation; BFRI = Bangladesh Fisheries Research Institute; BNP = Bangladesh Nationalist Party; HFMAP = Hilsa Fishery Management Action Plan; BOBLME = Bay of Bengal Large Marine Ecosystem; IIED = International Institute for Environment and Development. Cyclones, famines and other natural disasters are not noted here because of their frequency.

[4] Implements the Marine Fisheries Rules, 1983.

[5] Based on the Protection and Conservation of Fish Act and Rules, 1950, amended in 1985 by the Protection and Conservation (Amendment) Ordinance, 1982.

3.3.2 Institutional drivers

Institutional framework

The institutional arrangement of fisheries management in Bangladesh is hierarchical (Fig. 3.2; Appendix A.1). The Department of Fisheries (DoF), within the Ministry of Fisheries and Livestock (MoFL), is the principal organisation for the management and development of fish resources, represented at divisional, district and *upazila* (sub-district) levels. DoF activities are supported by autonomous organisations under the administrative control of MoFL: the Bangladesh Fisheries Development Corporation (BFDC) was established to promote the fishing industry and develop landing and marketing facilities; and the Bangladesh Fisheries Research Institution (BFRI) conducts research. Both the DoF and BFRI has been criticised for a lack of human resources, and BFRI for a lack of coordination between research bodies (DoF 2002; Islam et al. 2016).

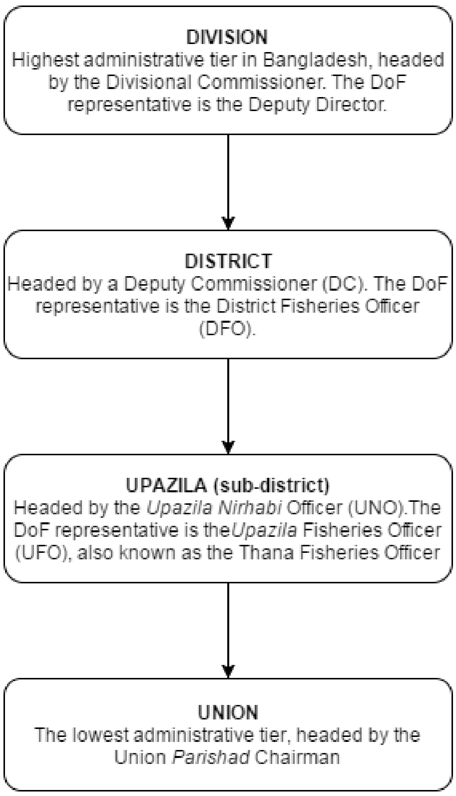


Figure 3.2: Administrative hierarchy in Bangladesh. DoF = Department of Fisheries

National fishery legislation

Of the 53 most active fishing countries in the world, Bangladesh has been ranked 47th based on its lack of compliance with the UN Code of Conduct for Responsible Fisheries (Pitcher et al. 2009), and the effectiveness of its marine fisheries management regime has been scored among the lowest in the world (Mora et al. 2009). In 1983 the DoF initiated a catch assessment survey system for inland and marine fisheries in Bangladesh, with particular emphasis on hilsa. There are, however, problems with sample sizes and sampling procedure; the assessment focuses on sample villages (many of which no longer exist) in major rivers and some marine areas, and it cannot therefore provide an accurate picture of catches (DoF 2002; Rahman et al. 2012b; Ullah et al. 2014). The creation of the marine wing of the DoF in the 1980s led to some improvements in the assessment of marine fisheries (Ullah et al. 2014). The DoF also enacted the Marine Fisheries Ordinance[6] and Rules[7] (MoFL 1983), under which waters less than 40 m in depth at high tide are reserved for the artisanal fishery and waters beyond this point are for the industrial fishery – although trawlers continue to fish up to 20 m depth (Islam 2003; Ali et al. 2010). All mechanised and, since 2001, non-mechanised vessels in the marine fishery must pay registration fees when commissioned, as well as annual vessel and fishing license fees (Islam et al. 2016). There is a restriction on numbers of trawlers fishing in marine waters, but no effort restriction has been imposed on the artisanal fisheries, and although the river fisheries have been through various systems of short-term licensing, they are now classified as 'open access' (Rab 2009; Ali et al. 2010). These 'open-access' river fisheries are, however, under complex systems of customary or traditional rights held by investors, which are not well documented (A. Bladon 2015, personal observation; Dastidar 2009).

All coastal and inland waters are managed under the Protection and Conservation of Fish rules, 1985, in accordance with the Fish Protection and Conservation Act, 1950 (MoFL 1985). Monitoring and enforcement of these rules by the DoF is supported by the Navy, coast guard,

[6] An ordinance is a law passed by a local administration.
[7] Rules define the guidelines that must be followed for the successful implementation of an act, which is a law passed by Government.

police, Rapid Action Battalion (RAB), Air Force, Border Guard and local administrations. However, management has been criticised for corruption at various levels, and a lack of resources and general institutional weakness has contributed to low levels of compliance (Chapters 5 & 7; Islam 2003; Ali et al. 2010; Jentoft & Onyango 2010).

In 1986 the New Fisheries Management Policy emphasised tackling the overexploitation of resources and the inequality of fishing rights (Islam et al. 2016). Then in 1998 a National Fisheries Policy (NFP) was prepared with the objectives of increasing fish production and promoting economic growth through sustainable fisheries management and aquaculture in inland open water and the sea, while conserving biodiversity and ecological function and alleviating poverty (MoFL 1998). It did not, however, formulate a specific strategy for the management of the artisanal hilsa fishery.

Hilsa Fisheries Management

Government budget allocations to the DoF for hilsa management have increased from about USD 4.11 million in 1998-1999 to USD 23.11 million in 2014-2015 (Majumder et al. 2015a). The DoF responded to concerns about hilsa catches in 1991 by initiating research on the fishery, which led to international collaborations with the Australian Centre for International Agricultural Research (ACIAR) and the World Bank-DFID Fourth Fisheries Project (FFP). Policy directives and recommendations from these projects have largely been implemented through the Hilsa Fisheries Management Action Plan (HFMAP). More recently, research has been conducted on hilsa through a Darwin Initiative project led by the International Institute for Environment and Development (IIED) and the WorldFish-led Enhanced Coastal Fisheries (ECOFISH) project, which is supporting the DoF to develop an updated HFMAP. Based on the assumption that recruitment is compromised, and that this is a result of overfishing of juvenile hilsa (locally known as *jatka*) and spawning hilsa, the HFMAP aims to sustain and increase hilsa production, prevent loss of hilsa habitat and build the capacity of implementing organisations

(DoF 2002). The following key recommendations have been taken up by implementing agencies[8].

Protection of spawning hilsa

Spawning hilsa are protected with a ban on hilsa fishing throughout the country for 15 days of the perceived peak breeding season[9], with the aim of minimising disturbance to spawning and recruitment (see Chapter 5). Monitoring for compliance with this ban is targeted within a 7000 km^2 area that is thought to cover important spawning grounds (see Fig. 3.3). But inconsistencies in ban periods between years and locations indicate a lack of communication between scientists and authorities (Islam et al. 2016).

Implementation of Protection and Conservation of Fish Act and Rules, 1950

Under the Protection and Conservation of Fish Act and Rules, 1950[10] (MoFL 1985), special operations for *jatka* conservation ban all activities related to *jatka* (catching, transportation, marketing, selling and possession) between 1st November and 31st July across the country. *Jatka* was originally defined as a hilsa fish of less than 23 cm long, but this has recently been amended to 25 cm (Islam et al. 2016). Use of monofilament gillnets (*current jal*) under 4.5 cm mesh size was banned in 1988, and both the use and production[11] of those under 10 cm mesh size is now banned, although not strictly enforced (Islam et al. 2016). The Fish Rules are implemented by the DoF's *Upazila* Fishery Officers (UFOs), and more recently the Navy and Coast Guard in the main rivers, which are reported to have improved enforcement (DoF 2002). It is not currently financially or logistically possible to enforce rules nationwide, so enforcement is targeted to the 152 *upazilas* where *jatka* are distributed, and particularly those areas that are thought to be important nursery grounds. However, the Fish Act has numerous institutional

[8] Those that have not yet been implemented are not mentioned here, except the recommendations for regional hilsa management, which are only beginning to be explored.
[9] This period was originally five days before and five days after the full moon of *Bara purnima* in the Bengali month of *Ashvin*, which falls in October, but was in 2015 extended from 11 to 15 days: 3 days before and 11 days after the full moon.
[10] Amended in in 1985 by the Protection and Conservation Ordinance, 1982.
[11] Use was banned in 1988 but the 2002 ban on production, marketing, importation, and possession was blocked by producers until resolution by the High Court in 2005.

weaknesses, and there have been instances of social and political interference with implementation (Islam et al. 2016). For instance, the Mobile Court Ordinance, 2007, allows a magistrate to operate a mobile court to deal with offenses on site, but resources are still lacking to gather mobile courts in time to do so. An analysis of trends and issues in enforcement can be found in Chapter 5.

Hilsa sanctuaries

Under the amended Fish Act and Rules (MoFL 1985), four major nursery areas in the Meghna and Padma rivers were designated hilsa sanctuaries in 2005, with another designated in the inshore marine area in 2011 (Fig. 3.3). All fishing is banned in these areas during their perceived peak period of *jatka* presence: March to April[12] in all but the southernmost sanctuary (a 40 km stretch of the Andharmanik River where fishing is banned from November to January).

[12] The ban was originally from February to March.

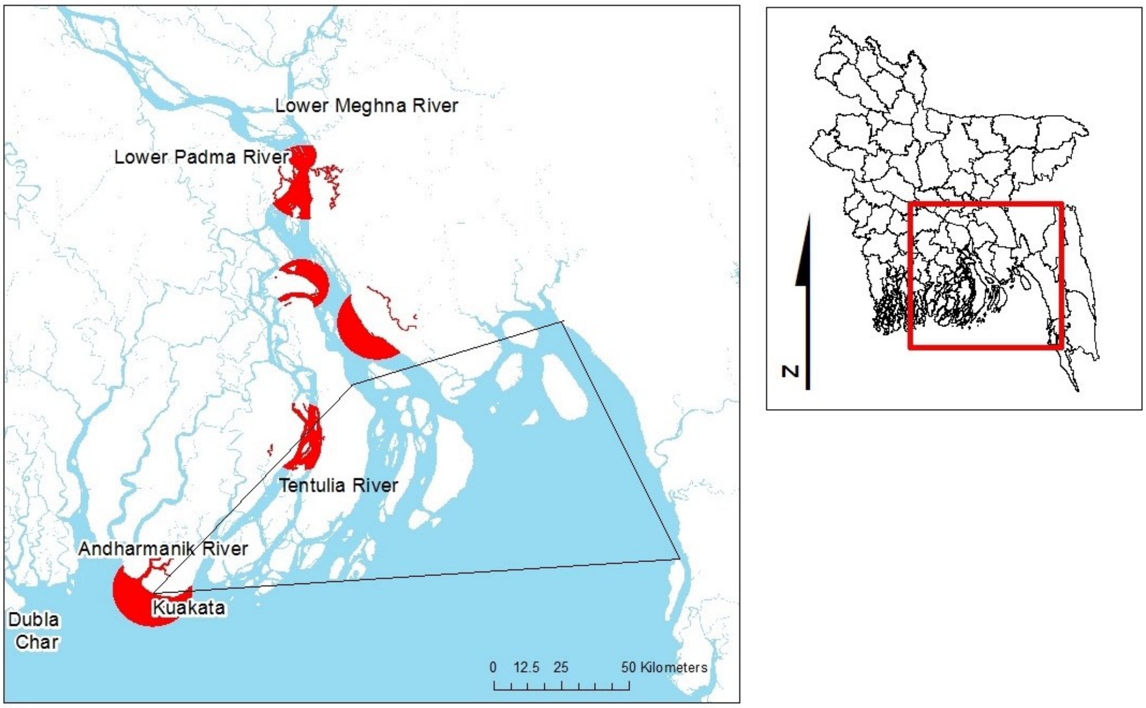


Figure 3.3: Map showing sanctuary areas (red) and rivers flowing into the Bay of Bengal (blue). From north to south the sanctuaries are: 100 km of Meghna River from Chandpur to Laxmipur, 20 km of Padma River in Shariatpur, 90 km of Shahbajpur channel (Meghna tributary), 100 km of the Tentulia River from Bhola to Patuakhali districts, and 40 km of the Andharmanik River in Patuakhali. All fishing is banned in the sanctuaries from March to April, apart from the Andharmanik River, where fishing is banned from November to January. Black polygon demarcates important spawning area where enforcement is targeted during peak spawning season.

Jatka fisher rehabilitation

In recognition of the socioeconomic hardships imposed by the fishing rules, in 2004 the DoF introduced the *jatka* fisher rehabilitation programme, which aims to improve the socioeconomic condition of affected fishers living inside and around sanctuary areas and thereby to incentivise compliance with the fishing bans (Islam et al. 2016). This is largely based on the distribution of compensation in the form of rice during ban periods (centered on the sanctuary fishing bans), which is funded through the pre-existing national Vulnerable Group Feeding (VGF) programme (Ahmed et al. 2009; Uraguchi 2011). Allocations and coverage increased from 10 kg per household for one to three months for 145,335 households in 2008, to 40 kg per household for four months (February to May, period of peak *jatka* abundance; Islam et al. 2016) for 224,102 households in 2014 (Table 3.1). However, the extent to which the food compensation actually incentivises compliance with regulations is probably limited by distributional issues and undermined by poor enforcement (Chapter 5; Siddique 2009; Haldar & Ali 2014; Islam et al. 2014).

In 2008 the rehabilitation programme was extended to provide a smaller proportion of households with alternative livelihood support such as rickshaws, vans, livestock and grants for small businesses. This support might be a more appropriate incentive than food distribution (Chapter 5) but, despite calls for increased coverage, the numbers of households receiving alternative livelihood support have actually decreased in recent years (Table 3.1), probably due to a lack of resources and needs assessment (Chapter 5; Siddique 2009; Islam et al. 2014).

Since 2003, awareness has been raised on the importance and status of the fishery, particularly *jatka*, and on the Fish Act and Rules, through boat rallies, mass media, and distribution of leaflets and posters (see Chapter 5).

Table 3.1: Distribution of rice compensation and alternative livelihood support (DoF 2014a). *Upazila* = subdistrict.

	RICE COMPENSATION				ALTERNATIVE LIVELIHOOD SUPPORT			
Year	**No. of *upazilas* (and districts)**	**Total volume distributed (mt)**	**No. of households**	**Monthly allocation household (kg)**	**No. of *upazilas* (and districts)**	**Total allocated amount (USD)**	**No. of households**	**Amount allocated per household (USD)**
2004-2005	-	1,000.00	-	-	-	-	-	-
2005-2006*	-	-	-	-	-	-	-	-
2006-2007	-	1,546.00	-	-	-	-	-	-
2007-2008	59 (10)	4,360.00	145,335	10	20 (4)	-	-	-
2008-2009	59 (10)	5,730.08	143,252	10	20 (4)	-	-	-
2009-2010	59 (10)	19,768.60	164,740	30	20 (4)	17,157.08	4,388	3.91
2010-2011	85 (15)	14,470.64	186,264	20	20 (4)	45,816.23	6,869	6.67
2011-2012	85 (15)	22,351.68	186,264	30	20 (4)	58,854.60	7,785	7.56
2012-2013	88 (16)	24,747.48	206,229	30	20 (4)	2,928.24	1,743	1.68
2013-2014	88 (15)	36,296.32	224,102	40	28 (6)	1,759.15	1,165	1.51

* Rice not provided

Marine fishing ban

In 2015 an amendment was made to the Marine Fisheries Ordinance and Rules (MoFL 1983), which bans all fishing by all vessels in the marine fisheries between May 20th and July 23rd each year, for conservation purposes. No science has been published to justify this ban, but the DoF has based it on concern for hilsa populations (A. Bladon 2016, personal observation.)

Regional hilsa management

Although Bangladesh shares its hilsa stocks with other countries in the Bay of Bengal (BoB), there is no formal agreement in place for regional management of hilsa (Rahman et al. 2012b). However, the Bay of Bengal Large Marine Ecosystem project (BOBLME) is supporting countries in the implementation of an ecosystem-based approach to the management of shared BoB fish stocks, under which a Hilsa Fisheries Assessment Working Group has been established to provide technical information to the BOBLME countries (BOBLME 2014). The IUCN has also proposed a set of policy options for transboundary management of hilsa for India and Bangladesh (Ahsan et al. 2014), and there has been a recent push to strengthen regional cooperation generally (BBS 2012).

Hilsa culture

As an alternative approach to increasing wild hilsa production, the MoFL has been supporting the exploration of pond culture (not found to be economically viable) and cage culture, which has had limited success (Puvanendran 2013; Sahoo et al. 2016). Captive breeding techniques are also being explored to supplement the natural population levels through 'hatch and release' (BOBLME 2014).

Summary

A number of acts, ordinances and rules have been passed that support the sustainable management of hilsa (Fig. 3.2), some of which are intended to support the artisanal hilsa fishery directly, largely through the protection of *jatka*. However, there are weaknesses in the institutional arrangements behind these policies, which lead to poor enforcement of rules and limit the effectiveness of the *jatka* fisher rehabilitation programme.

3.3.3 Economic drivers

Industrialisation

The economy of Bangladesh is rapidly developing (largely through industrialisation and exports) with an average national GDP growth rate of six per cent over the last decade (BBS 2012), and is classified as one of the 'Next Eleven' – a group of countries recognised for their potentially large, fast-growing markets (Goldman Sachs 2007). More than half of GDP is currently generated by the service sector, followed by industry (30 per cent) and by agriculture, forestry, and fisheries (16 per cent; BBS 2013). The garment industry, which emerged over the last two or three decades, provides the most foreign exchange earnings (80 per cent; CIA 2014). Other growing industries are shipbuilding and ship breaking; the world's largest ship-breaking area is in the Bangladesh city of Chittagong (BBS 2012). Over 45 per cent of the population is employed in the agricultural sector, but although there has been a substantial increase in food grain production in recent years, due to modernisation and mechanisation, agriculture is failing to absorb the rising labour force and its contribution to GDP is projected to fall (BBS 2012; GED 2012). Bangladesh has substantial untapped oil and gas reserves and is promoting international exploration (BBS 2012). Although it has limited coal reserves, there are also plans to increase coal-fired power generation (Allchin 2015).

Fishery sector

As a result of its geography, Bangladesh is heavily reliant on both inland and marine fisheries. They contribute over four per cent to the national GDP and are second only to the garment industry in foreign exchange earnings, which are derived largely from shrimp and prawn exports (FRSS 2013). Aquaculture is the dominant source of fish production (contributing over 50 per cent), while small-scale inland capture fisheries contribute about 30 per cent and marine capture fisheries 18 per cent (FRSS 2013). An estimated 88 per cent of marine catches can be attributed to artisanal fisheries, and the remainder to the industrial (trawl) sector. There has been a general upward trend in total fish production since 1983, but a steady increase in aquaculture production contrasts with a decline in capture fishery production since 2008 (Fig. 3.4; Table A.1). According to production statistics, the total fishery sector growth rate more than doubled between 2005 and 2012, although it should be noted that the reliability of these statistics is questionable. By way of example, a reconstruction of marine fisheries catches for Bangladesh from 1950-2010 found reconstructed catches to be 157 per cent higher than those reported by Bangladesh to the FAO, largely due to unreported subsistence catches (Ullah et al. 2014).

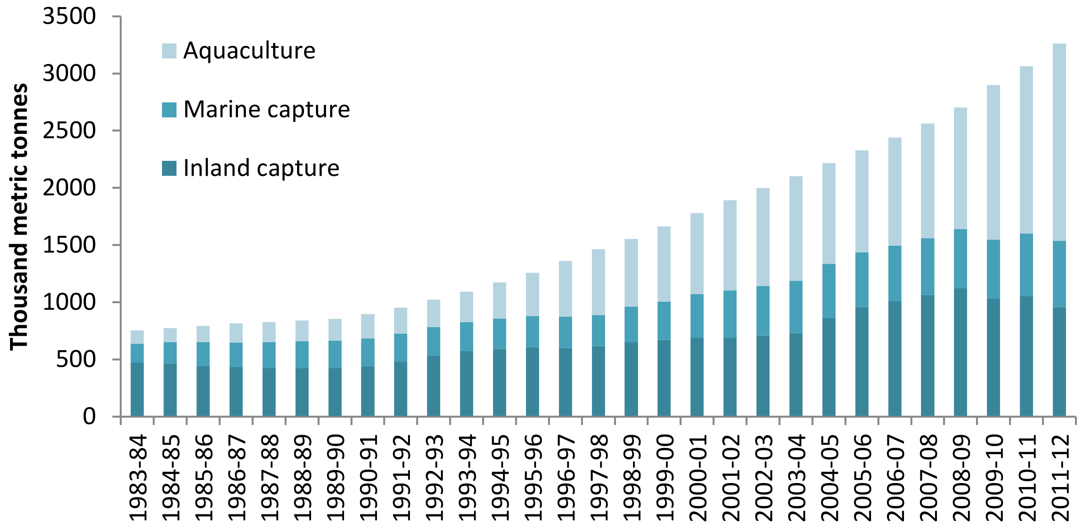


Figure 3.4: Graph showing the trend in total reported marine and inland fishery and aquaculture production in Bangladesh from 1983-84 to 2011-2012 (DoF 2014b).

Hilsa fishery

Bangladesh lands 50-60 per cent of reported global hilsa catches (BOBLME 2010). The hilsa fishery is a commercial one comprised of inland (artisanal) and marine (artisanal and industrial) sectors; hilsa are rarely caught purely for subsistence because they are high value fish (Ullah et al. 2014). There are large and highly variable numbers of fishing vessels in use, mostly unregistered: an estimated 100,000 vessels inland and 25,000 in the marine fishery (Rahman et al. 2010). Estimated numbers of hilsa fishers in Bangladesh range from 300,000 to 500,000 (Haldar 2004; Rahman et al. 2014a; Islam et al. 2016). Fishing occurs throughout the year, peaking from mid-August to October, when the majority of hilsa are reportedly caught (60-70 per cent), and to a lesser extent in January and February, inland (Rahman et al. 2012b). Fishing activity is reduced in most areas from November to July, particularly during the monsoon months from June to mid-August. Artisanal fishing practices are strongly influenced by traditional knowledge of lunar periodicity and tides passed down from generation to generation (Sharma et al. 2012).

Inland stocks are exploited using traditional non-mechanised boats (*chandi, khosa* and *dingi*), and some mechanised boats. According to the literature, the gears most commonly used are set gill nets (*chandi jal*) and drift gill nets (*gulti, kona* and *current jal*), with variable mesh sizes, but numerous others are in use (e.g. clasp nets, set bag nets and barrier nets; Rahman et al. 2012b). *Jatka* are targeted during their downstream migration, particularly at night when large numbers can be intercepted using illegal monofilament *current jal*, mosquito seine nets (*moshari jal*), set bag nets (*behundi jal*) and fixed encircling nets (*char ghera jal*; Siddique 2009; Rahman et al. 2010). Illegal *jatka* fishing is reported to have increased in recent years (Islam et al. 2016).

About 60 per cent of marine landings can be attributed to mechanised and non-mechanised boats and gill nets, and the rest to the industrial sub-sector that uses small trawlers to fish up to 250 km from the coast (Haldar 2004).

Traditionally, professional fishers in coastal areas were ‘low caste’ Hindus and fishing was considered taboo for Muslims, but now there are increasing numbers of poor and landless Muslims engaging in fishing, and wealthy Muslims investing in the fishing business (Hussain &

Hoq 2010). Artisanal fishers may fish from their own boat or a boat owned by a *mahajon*[13] (Alam et al. 2012). They often fish in groups of 4-20 to a boat, each fisher receiving small shares of the catch (Mome & Arnason 2007), whereas fishers working for industrial trawler companies can earn fixed salaries and catch-based bonuses (Kleih et al. 2003). Artisanal fishers often take *dadon* or loans for gear or boats from *aratders*[14] or *dadonar*[15], which commit the fishers to sell back to or through these middlemen exclusively and for lower than market price (Alam et al. 2012; Islam et al. 2016). Violence and piracy is widespread in the marine and inland fisheries (Kleih et al. 2003); pirates, known as *dacoits*, often have political connections and increase in numbers during peak hilsa season (R. Mohkles 2014, Centre for Natural Resource Management, personal communication, 23rd May).

The hilsa marketing system is long and complex (see Appendix A.2), with 2-2.5 million workers estimated to be involved in the market chain and ancillary activities such as boat-building, including women and children (Ahmed 2007; Mohammed & Wahab 2013). Fishers rarely sell catch directly to consumers, but instead operate through a number of intermediaries or middlemen (Alam 2010; Rahman et al. 2013). These intermediaries have complete control of the marketing system, which lacks Government regulation and exploits fishers (Islam 2003; Haque 2011; Islam et al. 2016). Not only does the presence of so many players in the market chain limit profitability for fishers, it locks extremely poor fishers in remote areas into cycles of debt, which traditional microfinance schemes have largely failed to drag them out of (Uraguchi & Mohammed 2016; Ahmed 2007; Ali et al. 2010; Alam 2012).

Most hilsa catch is marketed and consumed domestically as fresh fish (Alam 2010), but post-harvest management of fish in Bangladesh is generally poor. Only BFDC landings centres at Cox's Bazar and Chittagong (established specifically for the marketing of BFDC trawler catch)

[13] Relatively wealthy and powerful 'armchair fishers', who do not fish, but retain a large proportion of the catch or profit. Like other moneylenders, having given loans or credit in the form of boats or nets, they have complete control over fishing activities and market prices.
[14] Relatively wealthy and powerful commission agents who connect buyers and sellers of fish by handling the auctioning process at wholesale markets.
[15] Moneylenders, who can be *aratdars* or *mahajons*.

have adequate facilities (Islam 2003; Ahmed 2007), but there are about 6500 private fish markets in the country, more than half of which are small village markets (Rahman 1994, cited in Islam et al. 2016). Post-harvest loss rises during peak fishing season, when supply exceeds the availability of ice (see Appendix A.2). Until the late 2000s less than two per cent of total catch was legally exported to India and countries in the Middle East, Far East, Europe, USA and Australia where there are Bangladeshi diaspora, bringing in some foreign exchange earnings (Mome & Arnason 2007; Alam 2010; Table 3.2). However, the DoF has since implemented an export restriction (2007) and ban (2012 onwards) – a reported attempt to reduce the domestic price of hilsa and increase availability of the fish in national markets, with probable political motivations (Padiyar et al. 2012; M. Mome 2014, Department of Fisheries, personal communication, 29th May). In fact, a great deal is still being illegally exported to Kolkata (C. Meisner 2014, WorldFish, personal communication, 26th May).

Table 3.2: Officially reported hilsa export tonnage (metric tonnes) and earnings since 2002 (DoF 2014c). Unknown figures left blank.

Year	Hilsa export (mt)	Per cent of total hilsa catch exported	Export earning (million BDT)	Export earning (million USD)
2002-03	1148	0.6	150.00	1.91
2003-04	1930	0.8	790.00	10.07
2004-05	3584	1.3	519.50	6.62
2005-06	3672	1.3	696.10	8.87
2006-07	3433	1.3	648.10	8.26
2007-08	2647	0.9	754.50	9.62
2008-09	3680	1.2	1490.60	19.00
2009-10	3107	1.0	1241.20	15.82
2010-11	8539	2.5	3524.90	44.93
2011-12	6174	1.8	2940.00	37.48
2012-13	523*	-	243.70	3.11
2013-14	0	0	0.00	0.00

* Ban implemented half way through year

The reported contribution of hilsa to total reported capture fishery production has increased from 18 per cent in 1983 to 30 per cent in 2011 (Table A.1). Historically, fishing was concentrated upstream in major rivers, but the proportion of hilsa landings coming from the inland fishery has declined, and over the last two decades the marine sector has become the dominant source of hilsa (Fig. 3.5). This shift has been attributed both to *jatka* overfishing inland and to the mechanisation and expansion of the marine sector (Amin et al. 2002; Haldar 2004; BOBLME 2010). However, due to the previously discussed shortcomings of the catch assessment survey system, the reliability of the landings data is questionable. A reconstruction of marine catches from 1950 to 2010 indicates that hilsa makes up 18 per cent of total marine catches and 41 per cent of artisanal marine catches, and that reported landings have been underestimated by 19 per cent (Ullah et al. 2014).

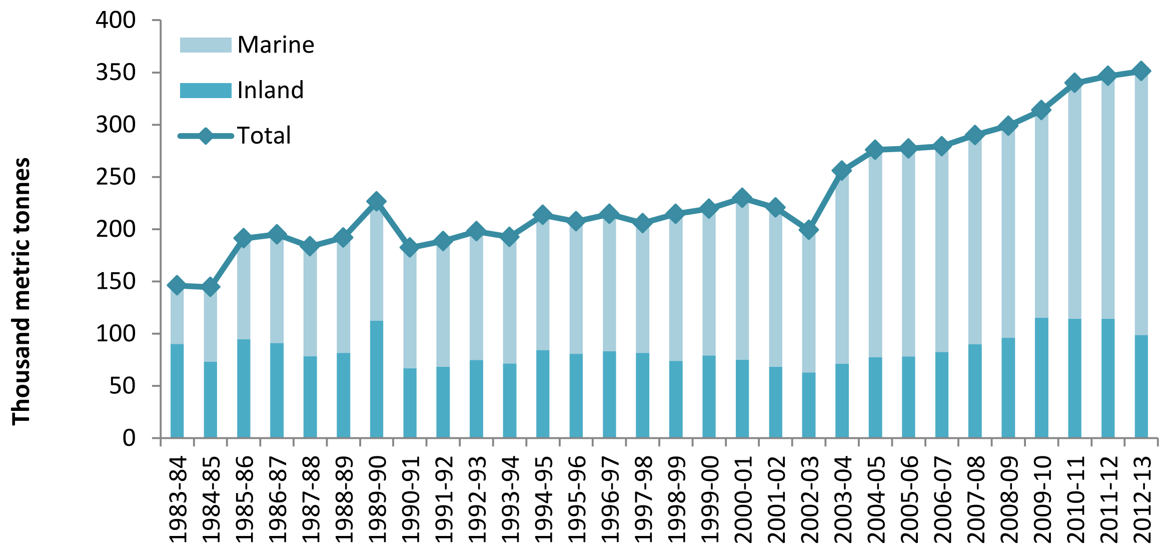


Figure 3.5: Total reported annual hilsa landings, inland hilsa landings and marine hilsa landings in Bangladesh from 1982-1983 to 2012-2013 (DoF 2014b).

The lack of reliable estimates of annual fishing effort also makes actual CPUE difficult to calculate, particularly for the inland fishery. Numbers of mechanised and non-mechanised boats in the artisanal marine sector have been estimated by the DoF and appear to have increased since 1983, which indicates that numbers of fishers (although not specifically hilsa fishers) are also increasing (Haldar 2004; Table A.2). Analysis of official marine landings and effort data shows a decline in CPUE between 1984 and 2006, although the trend was less clear for the non-mechanised sector (Mome & Arnason 2007; Sharma 2012). Sharma (2012) found catchability (the relationship between CPUE and standing biomass) of hilsa in the non-mechanised marine sector to be almost twice that of the mechanised marine sector – probably because the mechanised sector is targeting other species as well, rather than just targeting vulnerable concentrations of migrating hilsa, and has a shorter history of fishing.

Catch values are also difficult to estimate because of the complexity of the market chain and the relationships between fishers and intermediaries (Appendix A.2). Estimated annual landings revenues range from USD 380 million (or one per cent of GDP) in 2006 to USD 640-850 million in 2009 (Mome & Arnason 2007; BOBLME 2012). By multiplying an average value of 430 BDT per kg (based on a one year survey of four fish markets; Fernandes et al. 2015) by the latest reported annual hilsa catch volume, I found a much higher estimate of two billion USD. Through a very simple bio-economic assessment of the marine artisanal fishery (restricted by data limitations) Mome & Arnason (2007) found annual net profit to be only seven per cent of total revenues, due to high costs of fishing. Concluding that stocks were overexploited, they made recommendations for a 60 per cent reduction in effort, through restrictions on numbers of mechanised vessels in the artisanal marine sector. In turn this was expected to more than double hilsa stocks and increase individual vessel catch rates, which they calculated could raise annual net profits by 10-15 per cent. Yet, the industrial sector, in contrast to the artisanal sector, has been described as under-developed and under-utilised, and Bangladesh is now aiming for a 'blue economy' under which the trawler industry could expand (C. Meisner 2014, WorldFish, personal communication, 26th May).

Summary

Bangladesh is experiencing an upward trend in fishery production and hilsa forms an increasing proportion of this production. Landings and effort data suggests that this increase is largely due to expansion of the marine hilsa fishery, but that marine CPUE is nevertheless in decline. Yet, the reliability of these data is questionable. Moreover, very little is known about the inland fishery; inland landings appear to have remained fairly constant and clearly support the livelihoods of huge numbers of people, but inland fishing effort remains undocumented. Due to the absence of effort data, it is unclear whether the shift in dominance of production from the inland to marine hilsa fishery is due to inland overfishing, a decline in inland habitat quality or simply to expansion of the marine sector.

3.3.4 Social drivers

Human population

The population of Bangladesh has been growing since before independence, and it has one of the highest population densities in the world (UN 2015b). Despite this growth, there has been a steady decline in poverty over the last decade or so and two thirds of the population are now above the poverty threshold (World Bank 2013; UN 2015b). The Government's vision for 2021 is to bring it down to less than 15 per cent (GED 2012). The proportion of people living in urban areas increased from 15 per cent in 1980 to 30 per cent in 2015 and, since independence, rural-urban migration has accounted for two thirds of this urban growth (Afsar 2003; UN 2014). Extreme poverty is most prevalent in rural areas and poverty declines have been much more substantial in the west of Bangladesh than the east, where the hilsa fisheries are concentrated (World Bank 2013). The DoF estimates that fisheries support the livelihoods of 11 per cent of the total population (FRSS 2013) and fishers are often described as the poorest and most vulnerable social group in the country; they are usually landless with little education, low income and few or no other livelihood opportunities (Leterme et al. 2004; Deb & Haque 2011; Islam 2012). In Chapter 6, I demonstrate that low income is indeed associated with *jatka* fishing and with strong fishing dependence in hilsa areas, which is characterised by illegal fishing and lack of other livelihood activities.

Capture fisheries are not only essential for livelihood support but for direct consumption and dietary diversity, providing over 60 per cent of the animal protein in the Bangladeshi diet (FAO 2014; Belton et al. 2014). During the hilsa sanctuary fishing bans, malnutrition is a risk even for those households who receive rice compensation (Islam et al. 2016). Fish is a preferred food among Bengalis and hilsa are of great religious and cultural importance; it is sacred in Hindu mythology, features in numerous ceremonial and religious festivals, and has been named the national fish of Bangladesh (Sharma et al. 2012; Mohammed & Wahab 2013). Because supply does not meet demand, particularly during festival periods, market price is generally high (Padiyar et al. 2012) – although it was quite recently still one of the most widely consumed fish by all income groups (Belton et al. 2011). *Jatka* – being smaller and less tasty – has historically been the more affordable option for low income groups. These cultural values have bred a tradition of hilsa conservation – it is customary to buy a pair of hilsa on the day of *Vijay Dashami* (October) and not eat it again until *Basant Panchami* (February), a period which coincides roughly with the peak hilsa breeding season (Sharma et al. 2012).

Summary

Hilsa has tremendous social value, particularly for the coastal poor. The human population of Bangladesh is increasing, and poverty is declining – although much more slowly among hilsa fishers, where livelihood dependence on hilsa is high.

3.3.5 Physical drivers

Climate

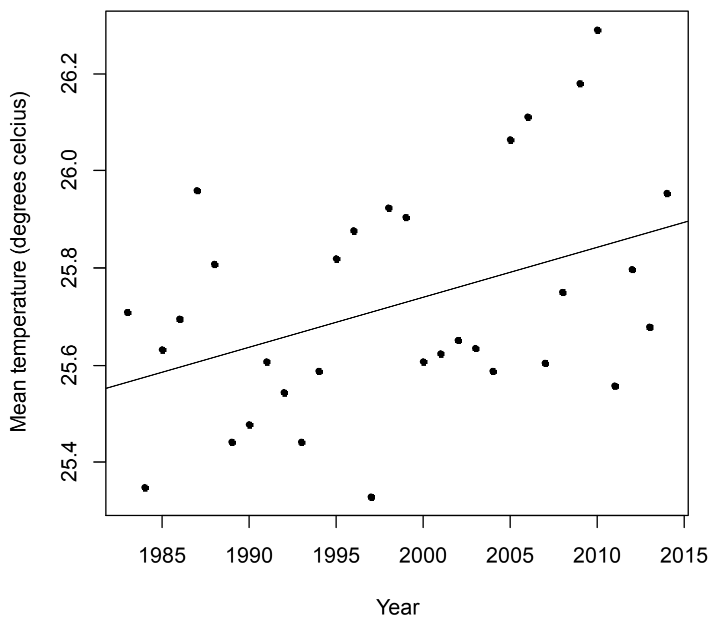
As a low-lying deltaic country, Bangladesh is extremely vulnerable to the impacts of climate change (Huq 2001). It suffers from intense tropical cyclones and storm surges and although there has been an emphasis on climate change adaptation research in recent years, the country lacks capacity for implementation and there have been few regional climate change studies focusing on Bangladesh (Agrawala et al. 2003; Reid & Huq 2014).

Within the study area of this thesis (Appendix B.1), over the period of 1983 to 2014, mean monthly temperature peaked in May at 28.9°C ±0.1, with a low of 18.6°C ±0.2 in January (Fig. A.3a). Mean monthly total rainfall showed even more variation, from a low of 6.9 mm ±2.4 in

January to a high of 922.3 mm ±43.0 in July (Fig. A.3b). This inter-annual variation is consistent with the main seasons; the dry winter season (December to February), pre-monsoon (March to May), monsoon (June to September) and post-monsoon (October to November) (Agrawala et al. 2003). Mean annual temperature in the study area has shown a significant upwards trend since 1983, whilst annual rainfall has declined significantly (Fig. 3.6a, Fig. 3.6b). The local trends in temperature are consistent with regional studies; Bangladesh as a whole has experienced warming over the last 100 years (Adger et al. 2003; Agrawala et al. 2003). Regional precipitation trends are unavailable but climate models tend to show an increase in precipitation during monsoon season – a trend which is consistent throughout South Asia (Agrawala et al. 2003; IPCC 2013).

There is no specific trend on which to base a regional scenario for sea-level rise due to the dynamic morphology of the Ganges delta, but approximately one fifth of Bangladesh lies within one metre of the high water mark (Huq 2001). It is therefore particularly vulnerable to the effects of the projected global rise in sea level, which in turn may compound the enhanced storm surges associated with cyclones, increasing flood risk (Karim & Mimura 2008). Salinity intrusion in coastal areas has also been linked both to the intensification of flooding (Dasgupta et al. 2014). It is unclear if or how these trends have affected hilsa, but given the importance of temperature, river flow and salinity in its life cycle, they are likely to have affected migratory and spawning behaviour, and ultimately production (Fernandes et al. 2015; Miah 2015).

(a)



(b)

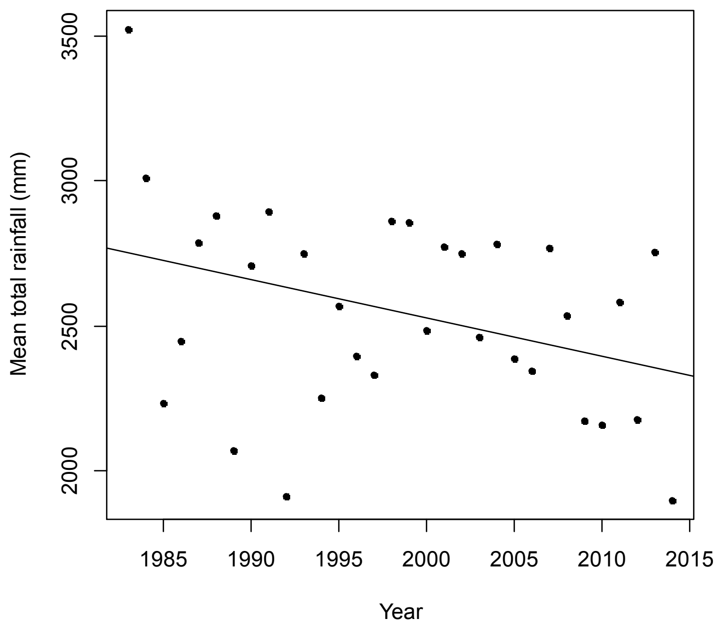


Figure 3.6: **(a)** Mean annual temperature from 5 meteorological stations in the study area, 1983-2014 (linear model: $r^2 = 0.169$, p = 0.019); **(b)** Mean total annual rainfall from 5 meteorological stations in the study area, 1983-2014 (linear model: $r^2 = 0.124$, p = 0.047). Source: Bangladesh Meterological Department (2014).

River hydrology and morphology

River channels are subject to constant changes in morphology through erosion and deposition. Annual monsoon inundations deposit alluvial sediments from the Himalayas into the Bangladesh river system (Fig. 3.7), which carries it into the BoB (Curray & Moore 1971; Shibly & Takewaka 2013), and the sandy islands that form as a result in downstream channels are thought to be blocking hilsa migratory routes (DoF 2002; Ahsan et al. 2014). High inter-annual and inter-study variability makes it difficult to draw conclusions on temporal trends in sediment loads entering the BoB (Islam et al. 1999).

The construction of dams and barrages for irrigation and flood control within and outside Bangladesh, together with estuary drainage and land reclamation projects, have also led to hydrological and morphological changes in rivers (DoF 2002; Ahsan et al. 2014). In particular, the construction of the Farakka Barrage on the Indian Ganges has led to a decrease in dry season freshwater discharge in Bangladesh (Mirza 1997), and branches and tributaries of the Padma and Brahmaputra – for example the Kumar River (Fig. 3.7) – are reported to have dried up as a result (Ahsan et al. 2014). By reducing the input of freshwater and silt, damming has also probably increased salinity downstream (Gupta et al. 2012; Shibly & Takewaka 2013). Some stretches of river are dredged to facilitate navigation and reduce flooding, and loop cutting – where the monsoon flood is diverted to eliminate meanders and loops, resulting in the deposition of large amounts of sediment – is a common practice in smaller rivers (Smith et al. 1998). Whereas dredging should open up migratory routes for hilsa, loop cutting may have the opposite effect. Many authors have linked these water diversion activities to hilsa population declines, and there are reports of large areas of hilsa spawning habitat having been lost in the upper region of the country due to these activities, particularly in the upper Padma and Kumar rivers (Blaber et al. 2003a).

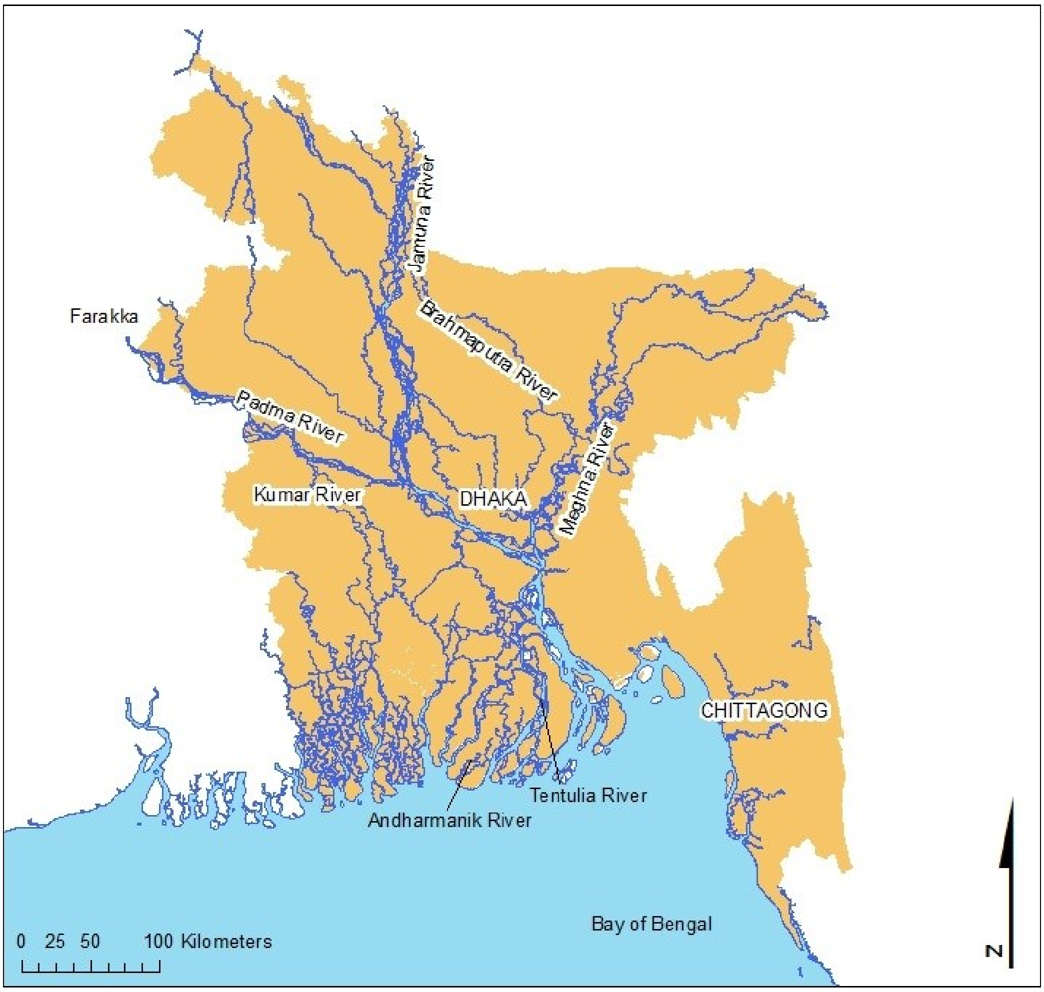


Figure 3.7: Map showing the major rivers of the Ganges-Padma river system in Bangladesh (Ganges-Brahmaputra delta). Where the Ganges flows out of India, its main channel becomes the Padma River, which joins the Meghna River, which continues to flow into the Bay of Bengal. Farakka marks the approximate site of the Farakka Barrage, across the border in India. Capital letters indicate major cities.

Deforestation

Forest cover is around 17 per cent in Bangladesh. Although this is an estimated 10 per cent of its original extent, recent net annual change appears to be stable or even positive (Laurance 2007; BBS 2012). Much of this cover is moist deciduous forest, but the south western coastal area is dominated by the Sundarbans mangroves and the freshwater swamp forests that lie behind them. Studies on forest cover have focused on this western coastal zone due to the important role mangroves play in shoreline stabilisation, storm protection, flood control and fishery support (Iftekhar & Islam 2004). Mangrove loss has been caused by timber collection, and by increased salinity resulting from unregulated encroachment of shrimp farming and storm surges (Hoq 2007). The importance of mangroves as fish habitat – particularly nursery grounds – and their impacts on fishery yields are clear (Aburto-Oropeza et al. 2008; Hutchinson et al. 2014). No direct link between forest cover and hilsa is known, and hilsa do not appear to favour mangrove areas. However, it is probable that hilsa are affected by changes in forest cover through nutrient availability and primary productivity.

Pollution

There is very little information on water pollution in Bangladesh and the BoB, but it is a clear environmental threat (BOBLME 2011a). Pollution is known to restrain phytoplankton growth and diversity, in turn reducing primary productivity (Huang et al. 2011). With few and poorly enforced environmental standards, industrial effluents and untreated municipal waste and sewage are released from urban areas into the river systems of Bangladesh, particularly around Dhaka (Fig. 3.7), eventually reaching the BoB (Karn & Harada 2001; BBS 2009; Hoque & Clarke 2013). Principal sources of inland pollution are tanneries, engineering, and pharmaceutical, textile, and chemical factories (Karn & Harada 2001). Water quality in polluted rivers has declined since 2000 (BOBLME 2011a), and there is evidence for a rapid increase in the production of hazardous industrial waste in Bangladesh between 1994 and 2007 (Waste Concern 2010). Eutrophication caused by the expansion of intensive agriculture and aquaculture activities is also an increasing problem (Islam 2003; BOBLME 2011a). Some areas of river are reportedly no longer suitable for hilsa as a result, particularly the Buriganga, which receives effluent from Dhaka (S.N. Chowdhury 2014, Winrock International, personal communication, 13th May), and possibly the Andharmanik (Chapter 5). In coastal waters, mechanised vessels, shipbuilding and ship breaking activities are producing petrochemical pollution, which has anecdotally been linked to reduced availability of hilsa in coastal areas (Das 2009). Petrochemicals can affect fish, their eggs and larvae and the plankton on which they feed, and can persist for long periods (Farrington 2014). Most research on marine pollution in Bangladesh has been focused on Chittagong (Fig. 3.7), a large port and ship breaking area (BOBLME 2011a).

Summary

Bangladesh is experiencing warming temperatures, a reduction in annual precipitation and probable intensification of monsoon precipitations, which could be compounded by sea-level rise. Sediment loads vary inter- and intra-annually, but water diversion activities are reducing inputs of freshwater and silt, and might be resulting in increased salinity downstream. Forest cover has declined dramatically but recently stabilised. Water pollution is an increasing threat both inland and in coastal areas, with potential impacts through direct mortality and reduction in productivity. Declines have been observed in adult and juvenile habitat quality and in river flows, which have been identified as attributes of susceptibility of the hilsa fishery to threat (BOBLME 2010), but the reported impacts of the trends presented here on hilsa are largely conjecture, and potentially difficult to tease apart.

3.3.6 Trends in hilsa

Although categorised as a species of least concern (Freyhof 2014), hilsa are – like many other shads – poorly understood, particularly in their marine phase (Chapter 5). Here I explore trends in abundance and distribution of hilsa by compiling all available information on the key rates that determine this abundance and distribution (movement, reproduction, growth, and mortality). I also explore trends in stock status; due to data limitations, no reliable stock assessments have been conducted in Bangladesh waters, but I review the information that is available.

Movement

Hilsa are distributed throughout coastal regions of the Indian Ocean, from Kuwait eastwards towards Myanmar (Whitehead 1985), but stocks in the Persian Gulf are genetically distinct from those in the Bay of Bengal (BoB; Milton & Chenery 2001; Salini et al. 2004). Hilsa are typically understood to be anadromous (migrating from marine to freshwater to spawn), but movements are complex and varied and there may be some permanent riverine and marine populations in Bangladesh (see Chapter 4 for detailed description of life cycle). Within the BoB, otolith microchemistry and allozyme variation provide evidence of substantial gene flow between groups of hilsa, indicating that fish in Bangladesh may also have spawned in India or Myanmar, and vice versa (Milton & Chenery 2001; Salini et al. 2004). Movements are strongly influenced by environmental conditions: water salinity, turbidity, temperature, pH, dissolved O^2, and phytoplankton availability – indicated by chlorophyll concentration or nutrient levels (Table 3.3). Water quality maps, generated using acceptable ranges of water quality parameters for hilsa, have been combined with catch data to explore its migration route through the Ganges-Padma river system (Ahsan et al. 2014). Upstream migration is associated with the increased flow and turbidity that comes with the monsoon, and fluctuations in hilsa abundance in different areas have been linked to monsoon intensity (BOBLME 2011b). The peak migration period is therefore generally understood to start somewhere between May and July, and to continue to October or November, although a shorter migration season in winter months from January to March has also been observed (Ahsan et al. 2014; Islam et al. 2016). Catch data from Bangladesh indicate that hilsa prefer water of at least 20 metres' depth for migration (Blaber et al. 2003a; Rahman et al. 2012a; Ahsan et al. 2014). Hilsa have been reported in the past to migrate over 1000 km upstream and major spawning grounds have been identified in and around the Moulvir char (about 120 km^2), Monpura (about 80 km^2), Khalirchar (about 194 km^2), and Dhalchar (about 125 km^2) areas of the Meghna River and Shahbazpur Channel (Fig. 3.3; Amin et al. 2004; Haldar 2004). Absence of hilsa has since been reported in some rivers and migratory distances are estimated to reach only 50-100 km, probably due to the disturbance of migratory routes (DoF 2002; Rahman et al. 2010; Sharma 2012; Miah 2015). *Jatka* can be found in all major rivers in Bangladesh, but major nursery grounds have been identified by experimental fishing in the lower reaches and estuary of the Meghna River and on the coast from Kuakata to Dubla Char (Fig. 3.3; Haldar 2004).

Table 3.3: Threshold values of physical and chemical parameters for hilsa spawning and nursery activities in Bangladesh (Ahsan et al. 2014).

	Spawning activities	**Nursery activities**
Depth	≥ 20 m for migration and pre-spawning congregation	Comparatively shallower depth
Turbidity (NTU)	100-140	70-80
Temperature (°C)	29.3-30.2	29.8-30.8
Salinity (ppt)	< 0.1	< 0.1
Dissolved O_2 (ppm)	5.0-6.8	4.8-6.8
pH	7.70-8.30	7.9-8.40
Chlorophyll (μg/l)	0.114-0.180	0.140-0.180

Reproduction

Fecundity varies with body size and habitat (Rahman et al. 2012a; Table 3.4), and although views on temporal trends are inconsistent (Blaber et al. 2003a; Haldar 2004; Miah 2015), historical data suggests an overall decrease in fecundity in the last four decades (Table 3.4).

Spawning (like migration) is influenced by exogenous factors (Table 3.3). Spawning occurs year round, with a peak in September and October following the monsoon flooding (Hasan et al. 2016; Rahman et al. 2012a; Ahsan et al. 2014; Bhaumik 2015).

Table 3.4: Historical hilsa fecundity data from 1968-2007 (Milton & Chenery 2001; Haldar 2004).

Date	Habitat	Length (cm)	Weight (g)	Egg numbers
1968	Padma-Meghna	22.5-48.3	-	900,000-2,000,000
1977	Meghna	38.0-52.0	-	382,702-1,821,420
1982	Padma-Meghna	33.0-51.0	-	600,000-1,500,000
1992	Padma (Goalunda)	26.6-51.1	228-1635	179,000-1,302,000
1998	Meghna	28.7-52.3	-	226,000-1,931,000
2001	Bangladesh	17.1-41.5	-	108,500-1,993,846
2002-2004	Ramgoti (Laxmipur)	35.5-47.0	448-1300	135,600-1,703,200
2002-2004	Kuakata (Patuakhali)	26.8-46.2	220-1270	209,000-1,088,200
2006-2007	Chandpur/Ramgoti	24.0-48.0	220-1130	112,554-950,625

Growth

From the point when the larvae become fry, hilsa feed on plankton (mainly phytoplankton) and so their growth is heavily influenced by phytoplankton availability, which can be estimated by measuring chlorophyll concentration (Table 3.3; Hasan et al. 2015). Growth parameters have been estimated based on length-frequency data collected between 1992 and 2009 (Table 3.5). Although these data are difficult to compare, because only one study adjusted data for gillnet selectivity (Rahman & Cowx 2008), they have been validated by other studies using length-at-age data based on otolith microchemistry (Milton & Chenery 2001; Blaber et al. 2003a; Rahman & Cowx 2008). A decrease in size at first capture can be seen between 1992 and 2000, indicating that many hilsa were being caught before reaching maturity, but it appears to have increased again since (Haldar 2004; Table 3.5).

Mortality

Total mortality comprises natural mortality (loss of stock through natural causes) and fishing mortality (removal of stock through fishing). Estimates of mortality parameters have been made from length-frequency data collected over time, though again it should be noted that methodologies differed between studies, making comparison difficult (Table 3.5). Estimated instantaneous rates of natural mortality are variable, ranging from a low of 0.98 in 1998 to highs of 1.36 in 2002 and 2009. Instantaneous rates of total mortality also peaked in 2002 at 3.51, compared with a low of 2.34 in 1998. Instantaneous rates of fishing mortality increased overall from 1992 to 2009, peaking in 1999 (2.49), and well exceeded natural mortality in every year. Fishing mortality was higher for inland samples than marine samples.

Hilsa landings data also provide some indication of trends in fishing mortality. Annual hilsa landings have increased since 1983 (Fig. 3.5). Hilsa are harvested throughout the year from the Padma River, the Meghna Estuary and the inshore waters of the BoB (FRSS 2013). Despite an upward trend in total landings, marine landings have actually increased at a much faster rate than inland landings, which have stayed fairly stable. However, as discussed in Section 3.3.3, marine CPUE appears to have declined.

Stock status

Stock assessments have been conducted in Bangladesh with hilsa population parameters estimated using length-frequency data (Amin et al. 2002, 2004, 2008; BOBLME 2010). Although this approach is not rigorous, it provides some of the only available indicators of hilsa abundance in Bangladesh and results are quite consistent. Between 1992 and 2009, estimated exploitation rate increased overall and fishing mortality rate was consistently higher than natural mortality, although since 1999 there has been a downward trend in exploitation and fishing mortality rate (Table 3.5). These studies broadly concluded that the fishery was overexploited, and attributed this to growth and recruitment overfishing, although some reports made contradictory recommendations for exploitation levels to be increased (DoF 2002;

Rahman et al. 2012a, 2012b). The only study that analysed marine and inland samples separately concluded that both populations were still under the maximum acceptable effort limit (Rahman & Cowx 2008). While inland hilsa stocks were found to be slightly overexploited, Rahman & Cowx (2008) concluded that the biologically optimal yield could still be obtained at a higher exploitation level for marine populations.

The BOBLME Project used a productivity-susceptibility analysis (PSA) to assess hilsa stocks in the BoB – a risk assessment approach with fewer data requirements than stock assessment, where 'productivity' is a composite measure of several key parameters (fecundity, catch rates, growth rates, age composition, mortality index and probability of breeding) and 'susceptibility' of attributes which determine susceptibility to threats (including protected areas, range and habitat quality; BOBLME 2010, 2011b). The study identified a declining trend in most of the productivity parameters in hilsa, but concluded that although there is evidence of recruitment overfishing in Bangladesh, stocks are not depleted or in need of rebuilding.

Systems dynamics simulation modelling has also been conducted using hilsa population parameters from the studies above (Bala et al. 2014). The study predicted a Maximum Sustainable Yield (MSY) of 268,000 tonnes (much lower than landings estimates in that time frame), and found growth rates of *jatka* and spawning adults to be very small, concluding that stocks are under 'severe stress' and vulnerable to overfishing. However, it did not show a decline in productivity; simulated weights of standing stock increased from 290,000 tonnes in 2004 to 380,000 tonnes in 2014 under current harvesting practice. These weights are higher than other estimates of 218,000 tonnes in 2003 (Mome & Arnason 2007) and an average of 95,144 tonnes from 1997-1999 (Amin et al. 2004).

Periodic surveys of experimental CPUE have been conducted in the Meghna river, and some authors noted a slight decline between 1998 and 2011 (BOBLME 2011b; Rahman et al. 2012b). Yet, these surveys shed no light on the status of marine populations, nor are they directly comparable. More recently, Sharma (2012) used time-series marine catch and effort data to estimate overall biomass with dynamic surplus production models and determined stock to be 15-30 per cent below optimal yield targets – 'marginally overfished to overfished.' Mome & Arnason (2007), on the other hand, conducted a bioeconomic assessment indicating that current estimates of marine effort were 33 per cent higher than the level of effort for MSY (Mome & Arnason 2007). Both of these studies used officially catch and effort data that are known to be unreliable (Ullah et al. 2014). The sensitivity of hilsa to exploitation is analysed in more depth in Chapter 4.

Summary

Hilsa range from the BoB to the rivers of Bangladesh, India and Myanmar, but this range may have been reduced. Understanding of migratory routes and the habitat types suitable for each life history stage is incomplete, and marine and inland populations have received limited individual research attention; whereas commercial CPUE data are available for marine populations but not inland populations, marine populations have received much less experimental research attention than inland populations. From the data available there appears to have been an increase in natural mortality, fishing mortality and exploitation rates, and a decline in size, maximum yield per recruit, and marine CPUE. However, much of this information is conflicting or limited in reliability, so trends in abundance are still unclear, and no data have been published beyond 2009. While some stock assessments have concluded that hilsa are overfished, exploitation rates are highly uncertain, and other risk assessments suggest that populations – although vulnerable to overfishing – may not yet be in decline.

Table 3.5: Growth, mortality, and exploitation parameters for the hilsa fishery in Bangladesh. Only in 1998** were samples from marine and inland waters analysed separately and adjusted for gillnet selectivity. '?' indicates unclear trend. Adapted from Sharma (2012)* and Rahman & Cowx (2008)**. Mortalities are instantaneous rates and thus can exceed a value of one.

	1992*	1995*	1996*	1997*	1998*	1998**			1999*	2000*	2002*	2003*	2009*	Overall trend
						Inland	Marine	Mean						
Asymptotic length (l_∞ in cm)	61.10	58.30	60.00	61.50	66.00	58.80	61.00	59.90	60.00	62.50	53.70	54.60	53.00	-?
Growth constant (k)	0.74	0.74	0.99	0.83	0.67	0.82	0.80	0.81	0.82	0.72	0.86	0.67	0.83	+?
Growth performance index	-	3.40	3.55	3.46	3.46	3.45	3.47	3.46	3.47	3.45	3.40	3.30	3.37	?
Length at first capture (l_c in cm)	35.0	30.0	30.3	29.81	27.06	-	-	-	22.80	13.12	19.87	21.21	26.00	-
Total mortality (Z)	2.41	2.61	3.19	3.29	3.43	2.38	2.30	2.34	3.77	2.79	3.51	3.07	3.23	+?
Natural mortality (M)	1.16	1.18	1.41	1.28	1.25	1.00	0.98	0.99	1.28	1.17	1.36	1.15	1.36	+
Fishing mortality (F)	1.25	1.43	1.78	2.01	2.18	1.38	1.32	1.35	2.49	1.62	2.16	1.92	1.87	+
Exploitation rate (E)	0.52	0.55	0.56	0.61	0.63	0.58	0.57	0.58	0.66	0.58	0.61	0.62	0.58	+
Maximum yield per recruit (E_{max})	-	-	0.71	0.69	0.60	0.61	0.65	0.63	0.59	0.46	0.58	0.63	0.57	-

3.3.7 The frame of reference

Having considered the above components in isolation, I can now combine them into a frame of reference. The data and understanding collated above constitute the baseline for the study – one of a complex, understudied and vulnerable fishery in a rapidly developing country, which suffers from technical and institutional gaps and numerous environmental threats (Table 3.6). I now analyse the trends and potential interactions between factors key to hilsa conservation interventions in Bangladesh (Fig. 3.8) to identify a set of potential counterfactuals (i.e., projected trends under positive and negative scenarios; Table 3.7).

Interactions

Current understanding of hilsa ecology indicates that any fluctuation in hilsa populations could result from a combination of institutional, economic, social and physical factors, which may compound each other and interact in myriad ways (Fig. 3.8). Aside from fishing, key drivers of change in hilsa appear to be the physical factors which interact to determine juvenile and adult habitat quality and availability, both of which appear to be declining (BOBLME 2010). For example, climate change, water diversion activities and related siltation may interact to block migratory routes, and pollution may interact with deforestation, climate and water diversion activities to reduce primary production and water quality. Most of these factors have multiple routes of potential impact. For example, in addition to direct mortality or reduction in habitat quality, pollution could also undermine current management interventions – if sanctuary areas are polluted (and Chapter 5 provides evidence to suggest that the Andharmanik river sanctuary might be), then the efficacy of temporal closures will be severely reduced. Another example is the reduced freshwater discharge and increased siltation resulting from water diversion activities; at the same time as reducing spawning grounds and reducing the suitability of habitat in terms of depth, salinity and turbidity, it may be concentrating migrations, making hilsa more vulnerable to being caught (Miao et al. 2010; Rahman & Bhaumik 2012a). While these diversion activities may reduce downstream siltation in some areas, sediment loading is closely coupled with monsoon inundation, so it is possible that intensification of this monsoon could be contributing to increased sediment loads elsewhere, reducing habitat suitability for hilsa (SRDI 1998; Goodbred & Kuehl 2000). On the other hand, dredging activities could be reopening some river channels for migration.

There are a number of positive and negative feedback loops at work involving socioeconomic change. In addition to direct impacts of climate change on hilsa feeding, spawning, and migratory behaviours, via changes in chemical and physical parameters of habitat, climate change could negatively impact hilsa populations via an increase in human vulnerability and poverty and therefore dependence on *jatka* in coastal areas (Adger et al. 2003; Dasgupta et al. 2014), or via an influence on water diversion activities. However, positive impacts may also arise through urbanisation and human migration away from coastal areas, which could reduce artisanal fishing pressure (BBS 2009; GED 2012). With urbanisation and industrialisation will probably come more livelihood options for hilsa fishers, although they may not necessarily be able or interested enough to adopt them (Wright et al. 2016). Although industrialisation has led to a reduction in poverty, this does not seem to have led to a reduction in numbers of artisanal hilsa fishers. In fact, increased mechanisation of vessels may have increased pressure on the fishery, and with urbanisation and industrialisation there comes pollution. A rise in the numbers of vessels and oil spills on the coast is causing petrochemical pollution (Rahman 2006), and as the recent oil spill on the south west coast of Bangladesh demonstrated, the country has little contingency for such disasters (BOBLME 2011a).

The interactions between hilsa fishing, institutional context and ecological trends are not discussed in depth here because this is the focus of Chapter 5. If the rationale underpinning current hilsa management and its implementation is sound, it should in theory protect against the overfishing of *jatka* and adults and thereby influence trends in hilsa abundance and distribution. However, the scientific basis for current management is often weak and there is evidence to suggest that implementation suffers from enforcement and compliance issues

(Chapter 5). Depending on its trajectory, fisheries management has the potential to either mitigate or exacerbate the impacts of climate change on hilsa (Fernandes et al. 2015).

Table 3.6: Trends and current baseline conditions for hilsa in the context of drivers of change and recent history (bioeconomic drivers refer to hilsa fishing and are discussed within Section 3.3.3).

Driver	Trends over recent history	Current baseline
Institutional	• Introduction of various management measures to increase hilsa production and protect habitat, with a focus on *jatka* conservation.	• Management by state institutions • Poor monitoring and enforcement of fishing regulations • Biological justification for management limited by lack of reliable stock assessment • No international regional management
Economic	• Rapid industrialisation and urbanisation	• Uncertain impacts, but pollution is probably affecting hilsa populations
Bioeconomic	• Contribution of hilsa to total fishery production has increased, largely through an increase in marine production • Marine CPUE has declined but inland CPUE is unclear • Foreign exchange earnings from hilsa reduced to zero due to export restrictions and ban	• Fishery has a low annual net profit • Production is dominated by marine sector • *Jatka* are caught mainly inland, often with monofilament nets
Social	• Human population has grown • Average poverty levels have declined, though more slowly in relevant regions	• Widespread illegal fishing activities linked to poverty • Hilsa have a strong cultural importance
Physical	• Habitat loss and degradation has disrupted migratory routes and led to a decline in range and abundance	• Hilsa prefer deep, clear, fast-running water with high phytoplankton availability • Majority migrate from marine to freshwater, but are not strictly anadromous

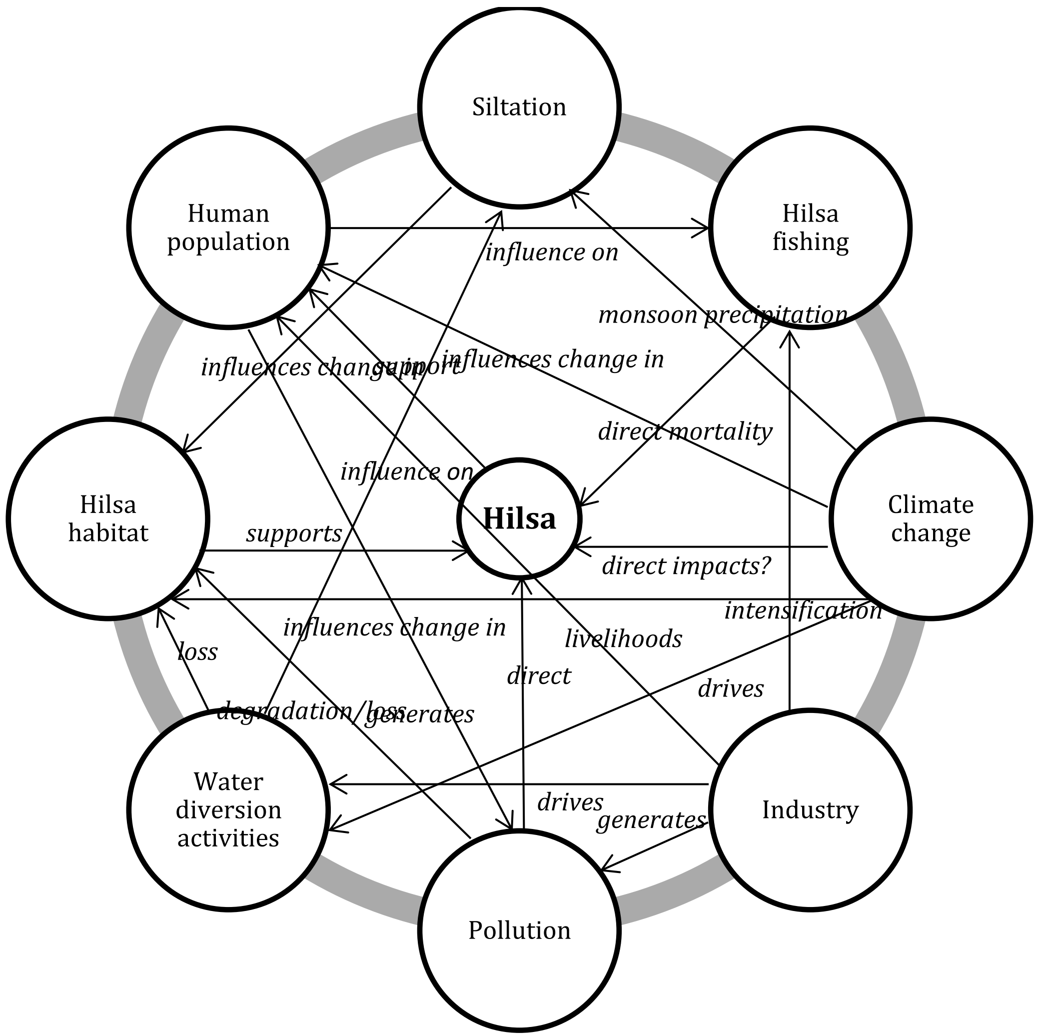


Figure 3.8: Conceptual map of potential factors key to hilsa conservation interventions in Bangladesh.

Counterfactuals

Bangladesh is likely to continue to experience climate change, with an overall reduction in precipitation and an increase in monsoon precipitation (Adger et al. 2003; Agrawala et al. 2003; Rahman et al. 2012c). Assessments of the potential impact of climate change on cyclone frequency and intensity in Bangladesh are tentative (Karim & Mimura 2008), but the IPCC has estimated a 5-10 per cent increase in peak intensity and a 20-30 per cent increase in associated precipitation rates (IPCC 2013), the effects of which would be compounded by projected global sea-level rise (Huq 2001; Karim & Mimura 2008). Model projections show a steady rise in sea surface temperatures in Bangladesh over the next century, with a potential increase in net primary production (Fernandes et al. 2015). Climate change is expected to cause a decline in fish production potential in South Asian countries that are highly dependent on fisheries, and specifically in Bangladesh (Barange et al. 2014; Habib et al. 2014). Modelling indicates that these changes will cause a decline in marine hilsa production potential in Bangladesh over the course of the century (Fernandes et al. 2015), although, within Bangladesh there may be some habitats that become more suitable for hilsa and others that become less suitable. Changes in water temperature, turbidity, nutrient levels and salinity are likely to impact migratory behaviour and therefore restrict hilsa distribution – especially when considered in combination with the possible effects of water diversion activities, associated siltation, land reclamation and pollution. Without habitat restoration, and depending on future rates of deforestation, submerged islands in downstream areas are likely to continue blocking migratory paths (DoF 2002) and spawning behaviour might also be affected by warming and changes in freshwater flow. Although marine and inland populations are inextricably linked, inland populations are assumed to be more vulnerable to the impacts of physical drivers and this is where depletion will likely occur first.

The human population of Bangladesh is projected to continue growing until around 2060 and the current majority rural population is expected to shift to majority urban (UN 2014, 2015). The Government's target for an annual economic growth rate of 10 per cent, based on accelerated growth in exports and remittances, would take Bangladesh across the middle income country threshold by 2021 and lead to a national decline in poverty (GED 2012; UN 2015b). Yet, it should be noted that continued political unrest and projected climate change (through increased frequency and intensity of natural disasters) could hinder economic growth (EIU 2015). Without the introduction and enforcement of more effective pollution prevention practices in Bangladesh, water pollution is expected to worsen with continued industrialisation and population growth (BBS 2012; Hoque & Clarke 2013). Rivers will be subject to increased waste generation, effluents from the garment industry – which is projected to continue growing (CIA 2014) – and eutrophication caused by the intensification of agriculture and aquaculture (BOBLME 2011a). Petrochemical pollution is expected to negatively impact hilsa populations in the BoB through increased numbers and continued mechanisation of vessels, growth of the shipbuilding and shipbreaking industries, and potential expansion of the oil and gas sector.

Together with economic drivers of urbanisation, projected changes in climate are expected to trigger continued migration of human populations away from coastal areas towards cities (BBS 2009). But climate change also increases poverty and vulnerability (Adger et al. 2003; Dasgupta et al. 2014) and, since rural coastal communities in the west of Bangladesh are expected to be the slowest to rise above poverty, dependence on fishing may remain high and *jatka* fishing may continue or even increase. These impacts will depend partly on local adaptation responses (Hobday et al. 2016).

In the absence of any institutional improvements, modelling suggests that increased levels of fishing effort on adult or *jatka* populations could lead to a collapse of hilsa stocks in Bangladesh within one to two decades (Mome & Arnason 2007; Sharma 2012; Bala et al. 2014; Fernandes et al. 2015). Given the Government's aim to increase hilsa production and given industry trends, the marine fishery – both artisanal and industrial sectors – looks set to continue expanding (DoF 2002). This expansion would require improvements in post-harvest technology and infrastructure, which could lead to overexploitation in the BoB (Habib et al. 2014). Even with these improvements, it is unlikely that the export ban will be lifted, and so expansion of the export market is unlikely. Total aquaculture production will almost certainly continue its upward trend, although the Government is mindful of maintaining its capture fisheries (Fernandes et al. 2015). Even if substantial progress is made with economically viable hilsa cage culture, it would be unlikely to take much pressure off the hilsa fishery, given the social and cultural importance of and demand for hilsa. Recommendations have been made for effort restrictions on the artisanal fishery, but given the weak enforcement capacity these are probably much less realistic than the current spatial and temporal fishing bans (Mome & Arnason 2007; BOBLME 2010). If a major threat to *jatka* and spawning adults is indeed overfishing, then improved monitoring and enforcement of current management rules could prevent collapse and potentially lead to increased hilsa abundance (Chapter 5). Sharma (2012) pointed out that since intrinsic growth rates are high, hilsa should respond quickly to conservation interventions. Projections indicate that with more sustainable management, some climate change impacts on marine hilsa production could be mitigated, although it would not halt a long-term decline (Fernandes et al. 2015). However, in light of the projected development of the marine fishery, the efficacy of these rules, which are enforced largely inland, is uncertain. The proposed development of a regional hilsa fishery management programme should reduce the impacts of physical drivers or fishing pressure from adjacent countries, but it is 'only an idea' (M. Khan 2014, Bangladesh Fisheries Research Institute, personal communication, 19th May).

If hilsa abundance does increase, this should in theory have a knock-on impact on profitability of the hilsa fishing industry. In turn, this profit could partially determine how much money is made available for hilsa conservation by the public or private sectors, although Mome & Arnason (2007) predicted that without substantial effort reductions, profits will remain low or even sink to zero. At the time of writing, a Conservation Trust Fund (CTF) was going through the ratification process, with the aim of generating financial resources for hilsa conservation (Majumder et al. 2015b). Although the CTF does not conform to best practice standards in terms of governance structure, it could be used to finance increased coverage of the *jatka* fisher rehabilitation programme and other conservation interventions, and pave the way for a more effective public-private partnership in the future (see Chapter 7).

Based on this evidence, a negative counterfactual for hilsa could be the collapse of hilsa populations in Bangladesh within one to two decades, due to a combination of overfishing, climate change and environmental change. But it is possible to develop an alternative positive counterfactual, where political will for institutional change, based on improved understanding of hilsa biology, effectively limits or reduces fishing pressure on hilsa populations, and mitigates some of the impacts of other anthropogenic activities, so that populations stabilise (Table 3.7).

Outstanding questions

Having developed two possible counterfactuals I highlight specific questions that would allow the most likely counterfactual to be established (Table 3.8). A key question is whether or not current management has the potential to protect hilsa populations. Due to the paucity of reliable time-series data, it is unclear whether the biological basis for these rules is sound – a question that will be explored in depth in Chapters 4 and 5. It is also yet to be seen whether *jatka* fisher compensation can incentivise compliance with rules, and whether the DoF will increase coverage or improve targeting effectiveness (Chapters 5 & 6). Further questions surround the extent of institutional change that might be introduced. The need for regional transboundary management, human resource development, new financing mechanisms, improved monitoring and enforcement, community-based management, equality of fishing rights, habitat restoration and adaptive management are all recognised in the HFMAP (DoF 2002), and although over a decade later most of these recommendations have not yet been implemented, areas in which progress have been made are financing and regional management. The creation of a CTF could be pivotal in terms of institutional change and sustainability of conservation interventions, but this will depend on the extent to which best practice is followed (see Chapter 7). In terms of fishing pressure, although poverty decline could result in a decline in illegal fishing activities, it should be noted that the social and cultural importance of hilsa and of fishing as a livelihood activity may counter any decline. Moreover, it is unclear whether urban economic development and industrialisation will actually drive down poverty in remote coastal areas. The potential impacts of climate change, polluting industries and water diversion activities are also uncertain, with implications for the appropriate placement of fishery closures, for the appropriate focus of conservation interventions, and for how adaptive they need to be.

Research needs

I now highlight key research needs for the hilsa fishery (Table 3.9). There is a clear need to improve hilsa stock assessment by conducting fishery-independent surveys of biomass on an international regional scale, in order to help managers understand the proportion of hilsa that should be protected for sufficient spawning biomass and which life stages to target. If the current catch assessment survey continues to be used, more data on age structure and size composition are required, and the survey should be extended from major rivers and estuarine areas to cover the whole country, including marine areas. Any assessments should employ a standardised modelling framework so that they can support a regional management plan. Collection of commercial catch and effort data should be extended to the inland fishery, if possible, where pressure on hilsa populations and the distribution of fishing activities are unclear, and a spatial analysis of fishing activities across all marine, riverine and estuarine areas would better allow optimisation of fishery closures. More ecosystem research and modelling are needed to establish the effects of potential physical drivers of change, particularly climate change, pollution and water diversion activities. Finally, a rigorous impact evaluation of management is required. The hilsa literature is full of claims of positive impacts since 2007, but none are convincing and their attribution to specific management interventions is not possible (Rahman & Bhaumik 2012b).

Table 3.7: Projected counterfactuals for hilsa in the context of drivers of change.

Driver	Negative counterfactual	Positive counterfactual
Institutional	• No effective change in institutional arrangements	• Protection of hilsa increases through improved monitoring and enforcement • Development of regional hilsa management plan • Fishery closures adapt to keep pace with environmental change
Social	• Poverty is slow to decline in coastal areas and illegal fishing continues, with no reduction in dependence on fishing	• *Jatka* fisher rehabilitation programme and job creation help to reduce illegal fishing and dependence on fishing
(Bio)economic	• Expansion of the artisanal fishery causes a decline in production and stock collapse within one decade	• Stable or reduced effort in the artisanal fishery slows a decline or stabilises production
	• Expansion of industrial fishery limited by lack of enforcement capacity	• Expansion of industrial hilsa fishery sustainable due to reduced artisanal fishing
	• No advances in hilsa cage culture or captive breeding	• Development of cage culture and captive breeding techniques reduces pressure on wild populations
	• Expansion of polluting industries	• Pollution prevention programmes mitigate some negative impacts of industry on hilsa in the long term
	• Climate change and political unrest limits economic development	• Economic development leads to job creation and urban migration
	• Existing power structures remain and continue to limit profitability to fishers	• Improved access to financial products increases profitability to fishers
Physical	• Water diversion activities, climate change, siltation and pollution disrupt migratory routes and reduce habitat quality	• Improved implementation of fisheries and environmental policies mitigates some disruption of migratory routes in the short term, but the long term impacts of climate change on habitat quality are unavoidable
	• Climate change may affect feeding, spawning and migratory behaviour via physical and chemical parameters	• No significant shift in behaviour is caused by climate change

Table 3.8: Outstanding areas of uncertainty relevant to establishing projected counterfactual, and their associated management implications.

Driver	**Uncertainty**	**Management implication**
Institutional	• Will institutional capacity be sufficient to maintain compliance with management rules?	• Determines whether fishing regulations and conservation payments can have impact
	• Do management rules have a sound biological basis?	• Determines whether management focused on the protection of *jatka* can have impact
	• Will a regional hilsa fishery management plan be developed?	• Determines whether management could be undermined by activities of other countries
	• To what extent will the Conservation Trust Fund follow best practice?	• Will affect sustainability of conservation interventions
Social	• Will poverty decline in coastal areas?	• May influence fishing dependence, illegal fishing activities, and therefore the appropriateness of conservation interventions
	• Will *jatka* fisher compensation incentivise compliance?	• Influences potential for ecological impact
(Bio)economic	• Will industrialisation provide employment and reduce poverty?	• May influence fishing pressure, particularly on *jatka*, and thus appropriateness conservation interventions
	• How much will the industrial hilsa fishery expand?	• Determines the level of coast guard enforcement required
	• Is hilsa overexploited?	• Determines requirement for effort control
	• Is *jatka* overexploited?	• Determines appropriateness of management focus
	• Will industry trends have a significant impact on hilsa populations?	• Determines whether interventions should focus on protecting habitat or controlling fishing pressure
Physical	• Will climate change block migratory routes and affect feeding, spawning or migratory behaviour?	• Determines whether interventions should focus on protecting entire migratory route or just the spawning grounds that remain, and how adaptive interventions need to be
	• Will water pollution have a significant impact on hilsa populations and where?	• Determines whether fishery closures will provide effective protection
	• Will siltation and water diversion activities block migratory routes?	• Determines whether interventions should focus on protecting entire migratory route or just the spawning grounds that remain, and how adaptive interventions need to be

Table 3.9: Key research needs for the hilsa fishery in Bangladesh.

Regional stock assessment	There is a need for regional fishery-independent estimates of spawning stock biomass and juvenile recruitment, which could be used to support an international regional fishery management plan. Age structure and size composition of Bangladesh hilsa populations would also help to develop current fishery-dependent assessments.
Improve catch and effort data collection	Catch assessment survey should be extended from the major rivers to cover the country more comprehensively. Currently commercial CPUE data is available only for the marine sector, but the inland sector provides about one third of estimated catch and so monitoring of these vessel numbers would shed light on the status of inland hilsa populations. Spatial analysis of actual fishing activities, as opposed to landings, would give a clearer picture of the fishery and could be used to optimise placement of fishery closures.
Impact of climate change on hilsa populations	Existing reports of climate change impacts on hilsa populations are largely anecdotal. A clear link and mechanism for change must be established and the types of habitats that should be protected for increased resilience should be explored.
Impact of water diversion activities on hilsa populations	Current reports of the impacts of damming on hilsa populations, though convincing, are still conjecture. Research linking quantitative habitat quality data to activities is needed.
Impact of deforestation on hilsa populations	Given the role that mangroves tend to play in fish production, this gap in research should be addressed.
Impact of pollution on hilsa populations	Quantitative studies of water quality, in relation to spawning and nursery areas, would help to ascertain whether pollution is undermining fishery closures.
Rigorous evaluation of current management, and in particular the rehabilitation programme	Currently there is no evidence to attribute any changes to either the fishery closures or any element of the *jatka* fisher rehabilitation programme, and a rigorous impact evaluation is required.
Aquaculture	Advances in captive brood-stock development, breeding and grow-out of hilsa may help to supplement or reduce pressure on wild hilsa populations.
Economic valuation of hilsa fishery	This could help to generate political will and investment in sustainable hilsa management.

3.4 Discussion

The task of developing and evaluating successful conservation interventions is always constrained by uncertainties arising from, for example, stochastic environmental variation, or from limited understanding of system dynamics – particularly resource user behaviour (Nicholson & Possingham 2007; Fulton et al. 2011). Environmental modelling approaches to dealing with this uncertainty are available (Refsgaard et al. 2007), but they have received limited attention in the conservation literature and are still rarely used in conservation decision-making (Nicholson & Possingham 2007; Milner-Gulland 2011, 2012). In fisheries, there is a widespread unwillingness among decision-makers to embrace uncertainty (Walters 2007). They have long expected from scientists single, clear predictions and management prescriptions from models parameterised with historical data, but the depth and complexity of understanding required for useful modelled predictions that reduce uncertainty to an acceptable level is often lacking, particularly in small-scale, developing-world fisheries. Learning from the failures and successes of weather and climate science, particularly the dangers of failing to effectively communicate uncertainties to decision-makers and other stakeholders, alternative approaches to incorporating uncertainty into fisheries management decisions are surely required (Clark 2001; Pidgeon & Fischhoff 2011; Davies 2015).

This study demonstrates how a useful frame of reference, comprising a baseline and counterfactual, can be developed and used to guide decision-making in complex, data-poor systems, even in the absence of reliable, predictable models. Similar in approach to scenario planning, this process can help to anticipate change, even when the projections themselves are based on incomplete information (Clark 2001). Although no quantitative counterfactual could be established for the hilsa fishery, the qualitative framework used here provides a way to break down all drivers of potential change in a system – institutional, social, economic and physical – and their complex interactions, reducing the potential for unexpected outcomes (Milner-Gulland 2012). The most likely counterfactual may not be an accurate prediction of the future, but critical evaluation of contradictions between studies and the reliability of understanding allowed the identification of key areas of uncertainty and their implications for management. This frame of reference thus provides a basis to explore the scope for potential additionality of current hilsa management (Chapter 5), and to consider the needs and opportunities for improvements (Chapter 7). Moreover, the identification of outcomes that were likely in more than one counterfactual scenario highlighted areas (e.g. climate change) that should be the focus of future research, if robust management interventions are to be designed and evaluated.

In conclusion, all fisheries are managed in the face of uncertainty, but management decisions should not be deferred by calls for further data collection or modelling when a reasonable reference frame can be developed. Building on the No Net Loss conservation policy literature in which the concept was developed (Bull et al. 2014, 2015, 2016; Bull 2014), this study is novel in its application of the frame of reference approach to fisheries. It is, however, in line with new thinking that affords uncertainty greater importance in fisheries management planning, even where model-based predictions are unreliable or impossible (Davies 2015). Analysis of this kind is rapid and inexpensive, and could thus be used on a regular basis to help guide an adaptive management approach to evaluation against a projected counterfactual (Bull 2014). Finally, following this framework is one way to ensure that uncertainty is considered from the very beginning of an intervention design process (Refsgaard et al. 2007).

Chapter 4

Characterising the impacts of selective fishing on the Bangladesh hilsa population

4.1 Introduction

Targeting very small and very large fish is common in developing-world fisheries. In small-scale, coastal fisheries, small and immature fish are often caught in easy-access intertidal nursery habitats such as mangroves and seagrass beds, and destructive or indiscriminate fishing gears are often used (Pomeroy et al. 2009; Hauzer et al. 2013). For instance, mosquito nets – the fine mesh of which entraps even very small fish – are widely used in malarial regions, particularly by fishers who lack the capital to invest in other fishing gear (Bush 2013; Gurung 2015). At the same time, many of these fisheries are open access and so may also be heavily overexploited (Mora et al. 2009; Purcell & Pomeroy 2015).

Targeting small and immature fish has long been associated with growth overfishing (Froese 2004); under yield-per-recruit (YPR) theory (Beverton & Holt 1957), which has dominated fisheries management discourse for decades, catching fish at sizes below their size at first maturity can reduce total YPR and therefore profit. The second form of overfishing – which interacts with growth overfishing – is recruitment overfishing, where spawning stock biomass is reduced to a level at which reproduction is substantially reduced. Under the spawn-at-least-once principle, allowing juveniles to mature and therefore reproduce should protect stocks from recruitment overfishing, even when exploitation rates are high (Myers & Mertz 1998). These impacts have been demonstrated in meta-analyses of empirical stocks and single species population modelling (Mori et al. 2001; Froese et al. 2008; Vasilakopoulos et al. 2011, 2015).

Measures to reduce the catch of small individuals – whether through fish size limits, mesh size limits, or other gear restrictions – are thus a focus of traditional fisheries management (Armstrong et al. 1990; MacLennan 1992; Halliday & Pinhorn 2002). These measures are particularly common in small-scale, artisanal fisheries because they are simpler, more easily monitored and often less controversial than measures to control effort, such as quotas and no-take zones (McClanahan & Mangi 2004). However, they have also been challenged from the perspectives of both traditional, single-species management and ecosystem-based fisheries management (EBFM).

Firstly, the assumption that all protected immature fish will contribute to future catches is rarely true; recruitment is heavily influenced by extrinsic factors such as environmental variability and the age range of the spawning population (Gilbert 1997; Cardinale & Arrhenius 2000; Szuwalski et al. 2015). Secondly, strong size selectivity can have evolutionary consequences. The selective removal of large, old individuals over small ones means that earlier maturation and smaller adult body size become evolutionarily advantageous, resulting in truncated age (size) structures, and potentially skewed sex ratios (Law 2000; Kuparinen et al. 2009; Law et al. 2013). Large, old spawners – often referred to as megaspawners or BOFFFs (big, old, fat, fecund females) – have long been recognised for their importance to healthy fish stocks. Large females are not only more fecund, but their eggs tend to be larger and thus hatch larvae with a greater chance of survival, and they tend to pass on genes for overall individual fitness (Longhurst 2002; Froese 2004; Hsieh et al. 2010). Mesh-size and gear restrictions therefore have potential to cause undesirable changes in the size structure of entire fish communities, which can in turn amplify temporal variations in biomass, reducing the resilience of populations to exploitation and environmental change (Anderson et al. 2008; Hsieh et al. 2010; Rochet & Benoit 2012; Kuparinen et al. 2016). This effect becomes particularly problematic at low abundance.

Furthermore, the rule that mesh size should be larger than a fish's size at first maturity was formulated for trawl fisheries, but gear selection curves differ between gear types: Wolff et al. (2015) recently demonstrated for gillnet fisheries that small mesh sizes promote sustainable production by allowing a high proportion of spawners to remain in the population. Not only do small fish grow more quickly through their window of vulnerability to being caught, but gillnets are passive gears and so have unimodal selectivity, i.e., small fish pass through the mesh and large fish rarely become entangled.

The selective fishing paradigm has also been questioned by proponents of EBFM (Garcia et al. 2012), since when trophic dynamics are taken into account, models suggest that the selection of larger fish can actually decrease total biomass yield, biodiversity loss and alter community structure (Pope 1991; Law et al. 2013, 2014). In single- and multi-species size-spectrum models, harvesting a particular size of fish will reduce predation mortality for smaller-sized fish, which tend to be more productive, and reduce prey availability for larger-sized fish (Law et al. 2013, 2014). These observations are particularly relevant for the developing world, where small fish play an important role in food security. The current focus of management on yields of large, high value but low productivity species is inefficient in terms of overall biomass and food security (Kolding & van Zwieten 2011; Charles et al. 2015; Kolding et al. 2015). The models have led to the concept of 'balanced harvesting', which proposes that moderate fishing across the widest possible range of species, stocks, and sizes in an ecosystem, in proportion to their natural productivity, should maintain relative size and species composition and thus increase total biomass and yields (Zhou et al. 2010; Garcia 2015). This theory can be observed in some real, small-scale, multi-gear and multi-species fisheries that use relatively unselective methods of fishing and yet maintain stocks (Kolding & van Zwieten 2011; Kolding & Van Zwieten 2014). Although it has come under heavy criticism as an unrealistic and even untenable strategy for management (Jacobsen et al. 2014; Charles et al. 2015; Froese et al. 2015; Pauly et al. 2016; Reid et al. 2016), the concept of balanced harvesting and its potential role in EBFM is still being debated (Andersen et al. 2016; Breen et al. 2016).

In the small-scale and largely artisanal Bangladesh hilsa (*Tenualosa ilisha*) fishery, both adults and juveniles – locally known as *jatka* – are harvested. Although adult hilsa are more desirable, *jatka* have historically been a more affordable and easily obtainable option for low income groups. Since mature hilsa mostly migrate upstream to spawn, *jatka* are targeted in inland and coastal areas during their downstream migrations, and can be caught in large volumes at night with small-meshed nets – the cost of which can be earned back in one night (Chapters 3 & 5).

Jatka fishing is therefore an activity that marginal farmers and labourers often switch to when income is very low (Haldar & Ali 2014).

Hilsa management is focused on sustaining and increasing production through the protection of spawning adults and, particularly, *jatka* (DoF 2002). Regulations include a) a seasonal ban on hilsa fishing for the protection of spawners; b) seasonal bans on all fishing in nursery grounds; c) a seasonal ban on all activities involving *jatka*; d) a complete ban on monofilament gillnets; and e) a rehabilitation scheme for hilsa fishers (rice compensation, alternative livelihood support and awareness-building activities) that aims to incentivise compliance with regulations for the protection of *jatka* (Chapter 5). This management approach is largely based on the normative argument that capturing juveniles will deplete stocks, although there are major gaps in the scientific evidence supporting the assumptions that underpin the approach. Not only are there insufficient data available for a reliable quantitative stock assessment, but the actual ecological impacts of different selectivity regimes on hilsa have not been studied. Moreover, the relative importance of selectivity and the exploitation rate in determining the sustainability of the fishery are unclear. It is therefore also unclear which fishers pose the most substantive ecological threat to the hilsa fishery: those targeting jatka or those fishing on larger individuals. In any case, there has been little documentation of the characteristics and activities of fishers in each sector.

Setting and meeting conservation and management objectives can be difficult when data are limited. However, rapid assessments of stocks and estimates of potential yields and biological reference points can be carried out using life history parameters (LHPs), which are much more available than survival and mortality estimates – or can be inferred from allometric relationships and related species (Beddington & Kirkwood 2005; Le Quesne & Jennings 2012; Zhou et al. 2012). For example, Beddington & Kirkwood (2005) developed techniques to estimate potential yields and sustainable fishery capacity directly from size and growth parameters and Beverton-Holt invariants, while Le Quesne & Jennings (2012) conducted a rapid risk assessment of fishing impacts on biodiversity in the Celtic Sea by assessing the sensitivity of species to fishing in relation to various conservation- and yield-based fishery reference points, using life history data.

In this chapter, I will a) provide context by characterising the Bangladesh hilsa fishery at the household level; b) use LHPs to model the potential relative ecological impact of different harvesting strategies on hilsa biomass at the overall fishery level; and c) model the spatial distribution of this ecological impact and its correlates across individual fishing households according to their reported harvesting strategies.

4.2 Methods

4.2.1 Household survey

Nine hundred fishing households were interviewed in a survey described in detail in Appendix B.1. Households were sampled from within and outside sanctuary areas, within the area where the compensation scheme operates, and from control sites outside the area where the scheme operates. Data used in this study came largely from the first two sections of the questionnaire (household characteristics and reported fishing activities; see Appendix B.1). I first characterised the hilsa fishery with a basic descriptive analysis of household fishing activities, using Chi-squared tests (or Fisher's exact where sample sizes were small), Wilcoxon signed rank tests and Wilcoxon rank sum tests. Since women do not engage in hilsa fishing (only processing, trading and other ancillary activities; Islam et al. 2016), I excluded 136 households with female respondents from the analysis. I also removed an outlier generated by a reported average catch volume that was far out of proportion to the rest of the distribution.

4.2.2 Modelling ecological impact

In order to assess the relative potential impacts of various size selectivity regimes on hilsa population biomass, I developed an age-structured population model with a Beverton-Holt stock recruitment (S-R) relationship, using LHP values taken from the literature (Fig. 4.1). I chose to measure impact in terms of equilibrium overall biomass rather than spawning stock biomass because, in largely artisanal developing-world fisheries, overall biomass is generally a management priority, and since this fishery targets juveniles as well as adults, spawning stock biomass is less relevant. To identify a fishing mortality (*F,* the instantaneous rate) reference point at which to assess the impacts of selectivity, I used yield-per-recruit (YPR) analysis – an approach that is useful when fishery selectivity data and growth and mortality parameters are available (Le Quesne & Jennings 2012; Zhou et al. 2012). I considered the F at which the maximum YPR (YPR_{max}) was achieved as a suitable reference point with which to explore ecological impact in the hilsa fishery (F_{max}), given the life history of hilsa (Section 4.2.3), the apparently high levels of exploitation in Bangladesh (Chapters 3 & 5), and the goal of maximising fisheries productivity (Chapter 5). But since evidence for overexploitation of the hilsa fishery is equivocal (Chapter 5), and taking into account the possibility that fishing at F_{max} can lead to recruitment limitation (Deriso 1982), I tested the sensitivity of the analysis to the assumption of F_{max} by assessing population biomass under half and double F_{max}. Under each of these three F scenarios, I then multiplied the relative ecological impact of each household's selectivity regime by their reported average catch volume to produce an overall index of potential relative ecological impact on hilsa of a household's fishing regime at a given value of overall F for the fishery as a whole. As a result of data restrictions and because the hilsa fishery is a large fishery of which the study households are only a small part, the household-level analysis of actual reported harvesting regimes was conducted in the context of separate assumptions made at the fishery level (i.e., that all households follow the same harvesting regime under a reference F).

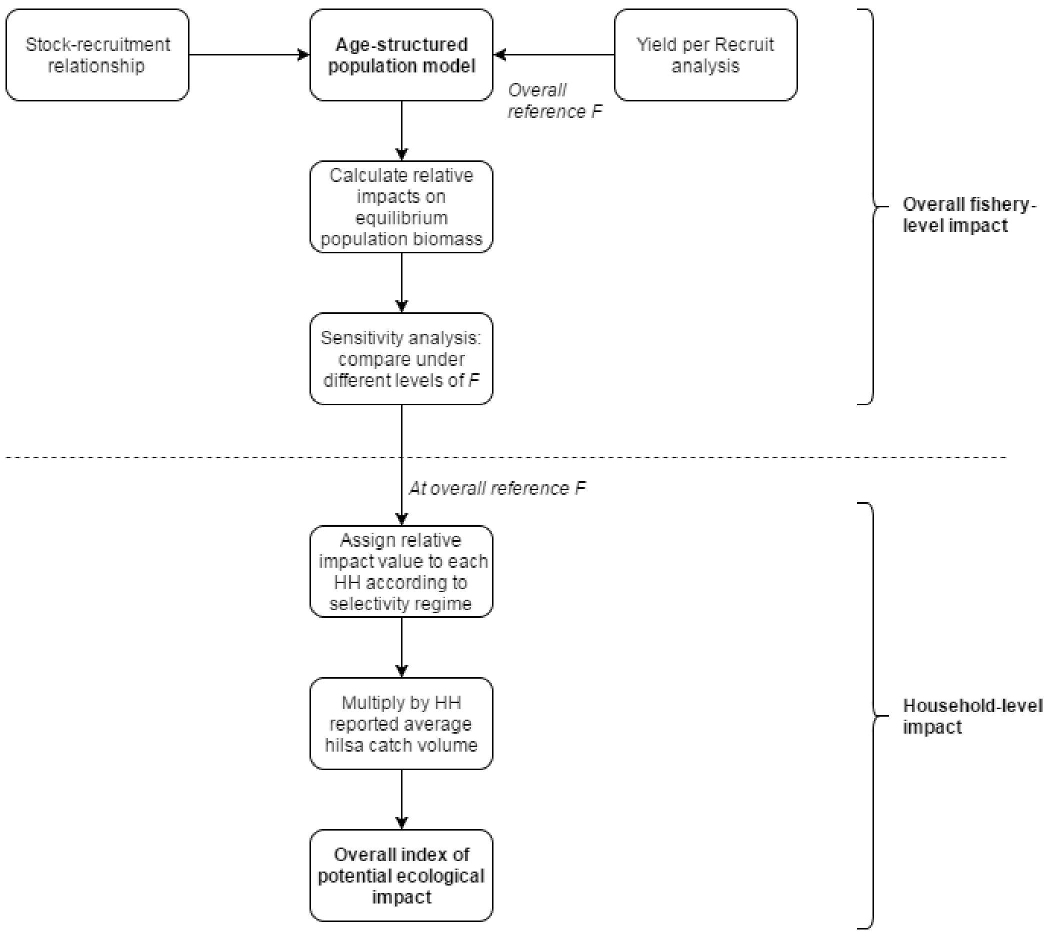


Figure 4.1: Flow diagram showing strategy for the development of a selectivity-based index of ecological impact on the hilsa population and for the development of a household (HH)-level index of potential ecological impact on the hilsa population, under the assumption of an overall reference fishing mortality (F).

4.2.3 Hilsa life history

It is typically understood that hilsa spend most of their lives feeding in the Bay of Bengal, with the majority of mature adults migrating in shoals upstream to spawn in freshwater (Fig. 4.2). Pelagic eggs hatch larvae within 23-26 hours of spawning, and after 6-10 weeks (12-20 cm) the fry become juveniles, locally known as *jatka* (Ahsan et al. 2014). These are officially defined as hilsa of up to 25 cm in length (Islam et al. 2014), but sizes at first maturity – which should be reached at about 1 year – range from 15 to 32 cm in the literature (Amin et al. 2000; Mome & Arnason 2007). After 5-6 months moving downstream, *jatka* continue migrating seawards to feed and mature, beginning the cycle again.

However, hilsa are not strictly anadromous; some spawning also occurs in estuarine waters, indicating that movements are complex and varied, and there may be some permanent riverine and marine populations (Blaber et al. 2003a; Rahman et al. 2010, 2012a; Bhaumik 2015). Spawning and migration are regulated not only by sexual maturity but by various physical and chemical factors (water depth, turbidity, pH, dissolved O_2, salinity and temperature; see Chapter 3). Although spawning occurs year round, peak upstream migration starts with the onset of the monsoon and associated flooding, with a peak in spawning in September and October (Hasan et al. 2016; Rahman et al. 2012a; Ahsan et al. 2014; Bhaumik 2015). There is some evidence to suggest that there may also be a distinct and smaller winter (January) spawning stock, which has a smaller migratory size and lower fecundity (Quddus 1982; Rahman et al. 2012b; Ahsan et al. 2014), but Hasan et al. (2016) challenge this. There is likewise some uncertainty about whether hilsa are iteroparous, or whether they demonstrate both iteroparity and semalparity (Blaber 2008; BOBLME 2011c).

Hilsa are very fecund; gravid females carry 0.1 million to 2 million eggs (Chapter 3; Rahman et al. 2012a), which together with their low age at maturation gives them a high intrinsic growth rate (r = 0.4 - 0.6; Sharma 2012). It is widely accepted that hilsa can live for up to 6.5 years, or about 58 cm in length – although larger fish have been recorded (Haldar 2004; Mome & Arnason 2007). Estimates of natural mortality from length-frequency data are highly variable with time and geography, ranging from M = 0.98 - 1.36 (see Chapter 3). An overall sex ratio of approximately 1:2 males to females is usually observed, with a bias towards females in larger fish (> 50 per cent) and towards males in smaller fish (> 50 per cent; Blaber et al. 2003a; Haldar 2004; Haldar & Amin 2005; Ahsan et al. 2014) – although some studies have identified populations with much higher female domination (Amin et al. 2005). There is no evidence of sex change.

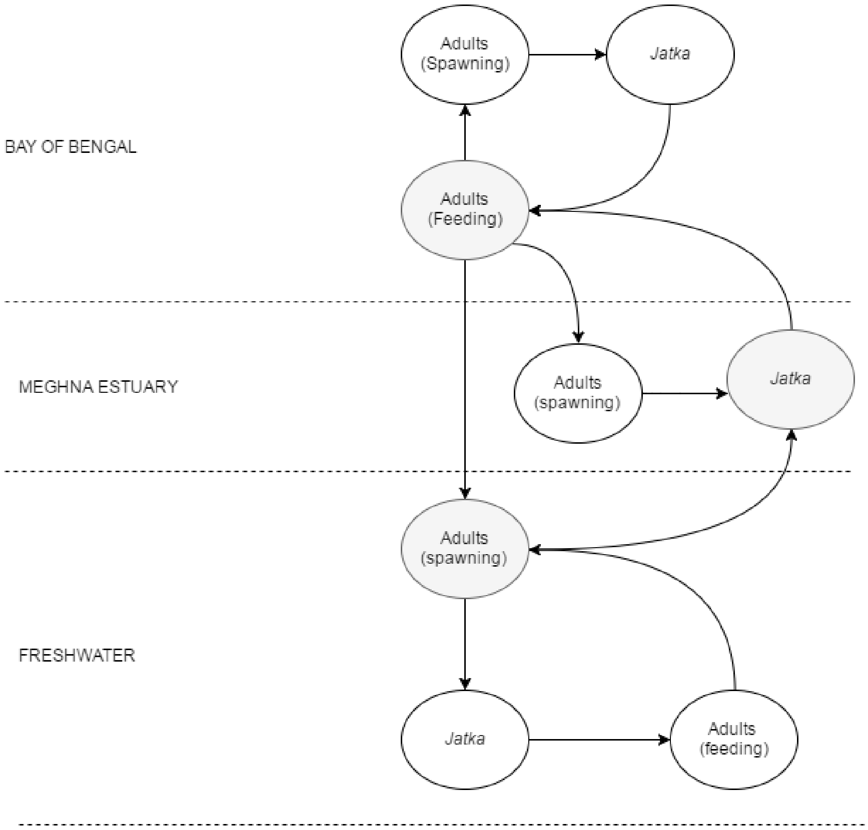


Figure 4.2: Migratory movements of hilsa (solid arrows indicate direction), adapted from Ahsan et al. (2014). The majority of hilsa migrate upstream to spawn, before returning to marine waters (route marked grey), but movements are complex and varied; there may be permanent riverine and marine populations.

4.2.4 Life history data

Since hilsa life history is not well understood and the availability of ecological data is limited (Chapter 3), translation of this life history into a population model was kept as simple as possible. Spawning seasonality and migration between marine and freshwater environments were not therefore incorporated. Although it varies between sources, length-at-age data supports the expectation that *jatka* are best represented in the model by fish < 1 year old (see Table 4.1). Fish were therefore assumed to mature at >= 1 year. The population model was structured by 6-month age classes up to 6.5 years; given hilsa's fast growth and high natural mortality rate, year classes were too crude to capture population dynamics (Mome & Arnason 2007).

Table 4.1: Length-at-age data for hilsa in Bangladesh (BFRI 2000), with the inferred stages of *jatka* (juveniles) and adults shown.

Age (years)	Length (cm)	Stage
0.5	15-16	*Jatka*
1	27-28	
1.5	36-37	Adults
2	42-43	
2.5	47-48	
3	51-52	
3.5	53-54	
4	54-55	
4.5	55-56	
5	56-56.5	
5.5	56-57.5	
6	57-57.25	
6.5	57.25-58	

I used LHP values taken from population dynamics studies of hilsa in Bangladesh, based on commercial catch length-frequency data (Table 4.2). When possible, I used values from the same study (Amin et al. 2004). Values for the length-weight constant (a_w) and exponent (b_w) ranged widely between literature sources (Amin et al. 2005; Ahsan et al. 2014) and those reported by FishBase (www.fishbase.org). However, substituting these values in and out did not affect final results, and so I used FishBase. Values for Beverton-Holt parameters and steepness could not be identified for hilsa. Therefore – and since there is much uncertainty regarding steepness – I used the median steepness for clupeids from a meta-analysis (Myers et al. 1999). This parameter characterises the steepness of the S-R relationship at low stock sizes (i.e., the recruitment obtained at 20 per cent of virgin biomass), and can be useful in assessing developing-country fisheries where the Beverton-Holt parameters are unavailable and difficult to estimate (Mace & Doonan 1988; Beddington & Kirkwood 2005). For virgin biomass I started with a recent landings estimate (351,223 t; DoF 2015), but having checked that the value makes little difference to the relationship between spawning stock and recruitment, I reduced the value to 2000 t for simplicity. The natural mortality parameter value was an average of instantaneous rates from three years of data (Amin et al. 2004).

Table 4.2: Secondary data used in this study, with sources.

Parameter	Value	Source
Steepness, h	0.71	Myers et al. (1999)
Asymptotic length, l_∞	62.5	Amin et al. (2004)
Growth coefficient, k	0.77*	Amin et al. (2004)
Length-weight constant a_w	0.0112	FishBase
Length-weight exponent b_w	3.0400	FishBase
Virgin biomass, B_0 (t)	2000	-
Instantaneous rate of natural mortality, M	1.27*	Amin et al. (2004)
Sex ratio (male:female)	1:2	IUCN (2014)
Proportion mature, r		Haldar (2004)
<=1 years	0	
>1 years	1	

*Halved from the values in original studies for application to half year classes

4.2.5 Household survey data

From the household survey the following data were relevant to the development of the ecological impact index (one data-point per household):

- Reported average catch volume (kg per fishing trip) in peak and lean fishing seasons;
- Reported size of fish in average catch (small/medium/large) in peak and lean fishing seasons;
- Reported presence of eggs/fry in average catch (many/few/none) in peak and lean fishing seasons;
- Whether or not the household targets *jatka.*

Peak fishing season is defined as the period from mid-August to October when the majority (60-70 per cent) of hilsa are reportedly caught (Rahman et al. 2012b). In lean season (November to July) relatively few hilsa are caught, particularly during the monsoon from June to mid-August when weather conditions are unfavourable. Reported average hilsa catch per fishing trip was significantly higher in peak season than in lean season (Wilcoxon signed rank test $V = 275653$, $p < 0.001$). Due to small sample sizes in some categories, reported size of fish and presence of eggs/fry in the catch could not be used as proxies for potential ecological impact in the development of the index. Instead, *jatka* fishing was used as a measure of selectivity, while catch volume was used as a proxy for household-level exploitation rate. Fifty-two per cent of households reported that they target *jatka*. There was no significant association between reports of targeting *jatka* and reported fish size in peak or lean season (Fisher's exact test, $p > 0.05$), which was to be expected, given that so few households said that they catch small fish (0.6 per cent in peak season and 2.3 per cent in lean season). This can be taken as evidence that respondents were not interpreting 'small' fish to be synonymous with *jatka* in this survey. Because *jatka* fishing is illegal, there may have been some strategic bias in whether a fisher self-identified as a *jatka* fisher, but respondents seemed willing to offer the information.

There were significant associations between reported fish size and egg/fry presence in catches (binary variable; 1 = many, 0 = few/none) in both fishing seasons. The majority of households who reported eggs/fry in peak season also reported catching large fish, but those who reported few or no eggs/fry reported catching smaller fish (Fisher's exact test, $p < 0.0001$). In lean season, reporting few or no eggs/fry was significantly associated with catching medium fish (Fisher's exact test, $p < 0.0001$). Although sample sizes were small (only 13 individuals reported eggs in their peak season catches), the association between reports of egg/fry presence in catches and peak fishing season provides some validation of the use of fisher knowledge in this study, given that peak spawning season does fall in peak fishing season (Chapter 5). Since larger fish are generally expected to be more fecund (Froese 2004), the association of reports of larger fish with egg/fry presence provides similar validation.

4.2.6 Model construction

Calculating the stock-recruitment relationship

The Beverton-Holt S-R relationship (Fig. 4.3) can be presented as follows, where S is spawning stock size, a is the maximum number of recruits produced and b is the spawning stock needed to produce recruitment, R, equal to half maximum:

$$R = \frac{aS}{b + S} \tag{1.1}$$

In the absence of a and b values, this equation was re-parameterised in terms of the steepness of the stock recruitment curve, h, the initial recruitment, R_0, and virgin biomass, B_0 (Haddon 2011). Assuming that recruitment in the virgin stock derives from the virgin biomass, then:

$$R_0 = \frac{aB_0}{b + B_0} \tag{2.2}$$

And:

$$h = \frac{0.2(b + B_0)}{b + 0.2B_0} \tag{3.3}$$

As estimates of growth parameters l_∞ (asymptotic length in cm) and k (growth coefficient) were available, I was able to use the von Bertalanffy (1938) growth equations to estimate average length at age, l_i in cm, and thus average weight at age, w_i in kg, where a_w is the length-weight constant and b_w is the exponent, and i is age class (see Table 4.2). Age, i, was taken to be the upper bound of each age class.

$$l_i = l_\infty[(1 - e^{-k})i - B_0] \tag{4.4}$$

$$w_i = a_w {l_i}^{b_w} \tag{5.5}$$

Given A_0, the mature biomass per recruit from the stable age distribution, where $n_{0,i}$ is the virgin number of fish per recruit at age i, w_i is weight at age i, m is the age at maturity (assumed to be equal to age of recruitment) and M_i is the instantaneous rate of natural mortality rate at age i, then R_0 can be calculated:

$$A_0 = \left(\sum_m n_{0,i} w_i\right) e^{-M_i} \tag{6.6}$$

$$R_0 = \frac{B_0}{A_0} \tag{7.7}$$

Parameters α and β can then be defined:

$$\alpha = \frac{B_0 \ (1 - h)}{4hR_0} \tag{8.8}$$

$$\beta = \frac{5h - 1}{4hR_0} \tag{9.9}$$

$$a = \frac{1}{\beta} \tag{10.10}$$

$$b = \frac{\alpha}{\beta} \tag{11.11}$$

The secondary empirical length- and weight-at-age data provide some validation for the estimates calculated in this study using the Beverton-Holt S-R relationship (Fig. 4.4a & b). The empirical data appear to be 3-6 months ahead of the estimates, with the largest differences in the mid-range years, but given that the empirical data were collected in just one year, the fit is quite good. The differences could be related to claims that size-at-first-capture has declined since the length and weight data were collected in 2000 (Haldar 2004), or to temporal variability in hilsa recruitment. The length-at-age data (Fig. 4.4a) are more similar to the model predictions than the weight-at-age data (Fig. 4.4b), which is probably because weight tends to be more environmentally determined than length.

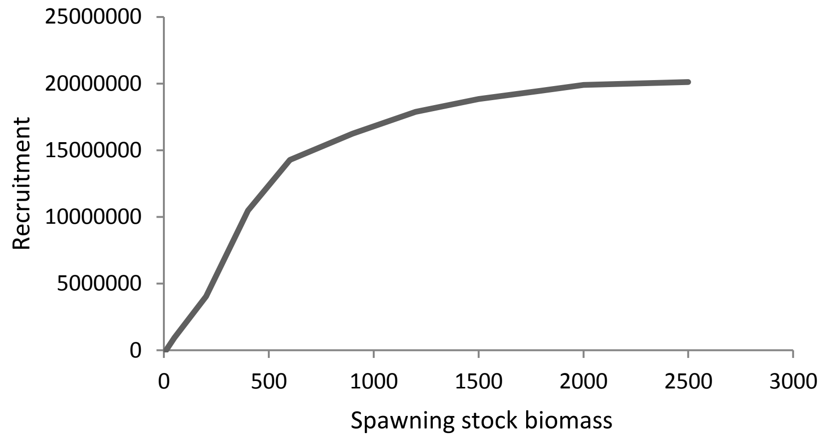


Figure 4.3: Beverton-Holt stock-recruitment relationship for hilsa, calculated using growth parameter data from the literature.

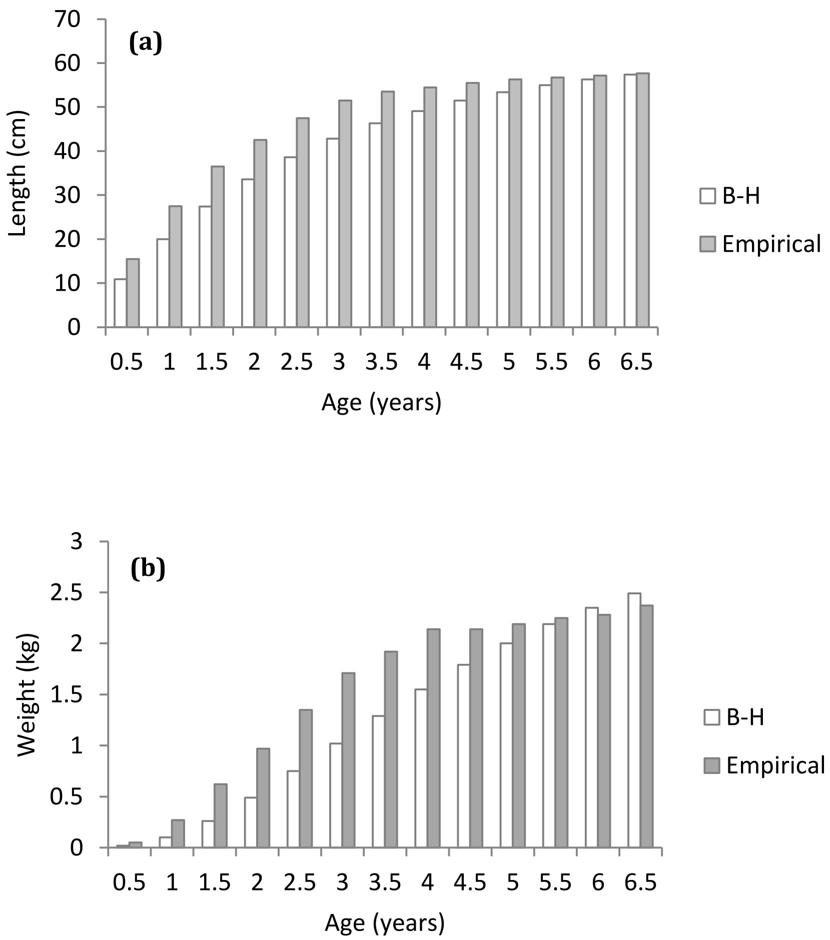


Figure 4.4: Histogram showing **(a)** length-at-age; and **(b)** weight-at-age estimated using the Beverton-Holt (B-H) stock-recruitment relationship, compared to empirical data collected during one year (BFRI 2000).

Calculating fishing mortality

In the absence of reliable estimates of fishing mortality, a reference point for the overall instantaneous rate of fishing mortality, F, at which to run the population model was identified through YPR analysis. The numbers at any age i+1 were calculated using the equation for exponential decline, where M_i is the instantaneous rate of natural mortality and F_i is the instantaneous rate of fishing mortality:

$$N_{i+1} = N_i e^{-(M_i+F_i)} \tag{2.1}$$

The numbers caught in a given age class, C_i, were calculated, where n_i is the number of fish from the cohort remaining in a given age class, M_i is the instantaneous rate of natural mortality in a given age class and F_s is the instantaneous rate of fishing mortality in selected age classes (where $F_s = 0$ in non-fished age classes).

$$C_i = \left(\frac{F_s}{M_i + F_s}\right) n_i \left(1 - e^{-(M_i+F_s)}\right) \tag{2.2}$$

Thus biomass yield, Y, from the cohort as a whole at time t, could then be calculated, where I is the maximum number of age classes and w_i is weight in a given age class:

$$Y = \sum_{i=I}^{I} w_i \, C_i \tag{2.3}$$

To generate YPR, Y was divided by initial cohort size, N_0:

$$YPR = \frac{Y}{N_0} \tag{2.4}$$

By running the YPR model under a range of values of F, I estimated the maximum yield obtained under non-selective fishing, Y_{max}, which corresponded to an F_{max} of 0.8 (Fig. 4.5).

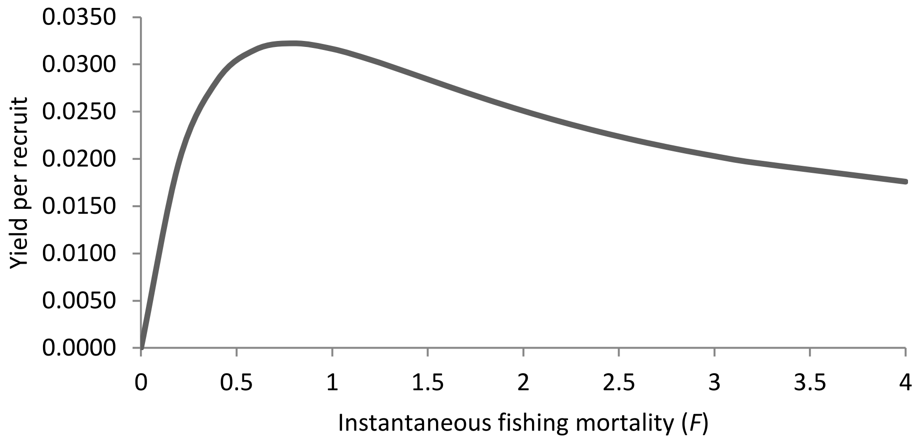


Figure 4.5: Yield-per-recruit curve for hilsa under a regime of fishing across all age classes.

The operating model

I followed the steps in Fig. 4.6 to develop the age-structured population model.

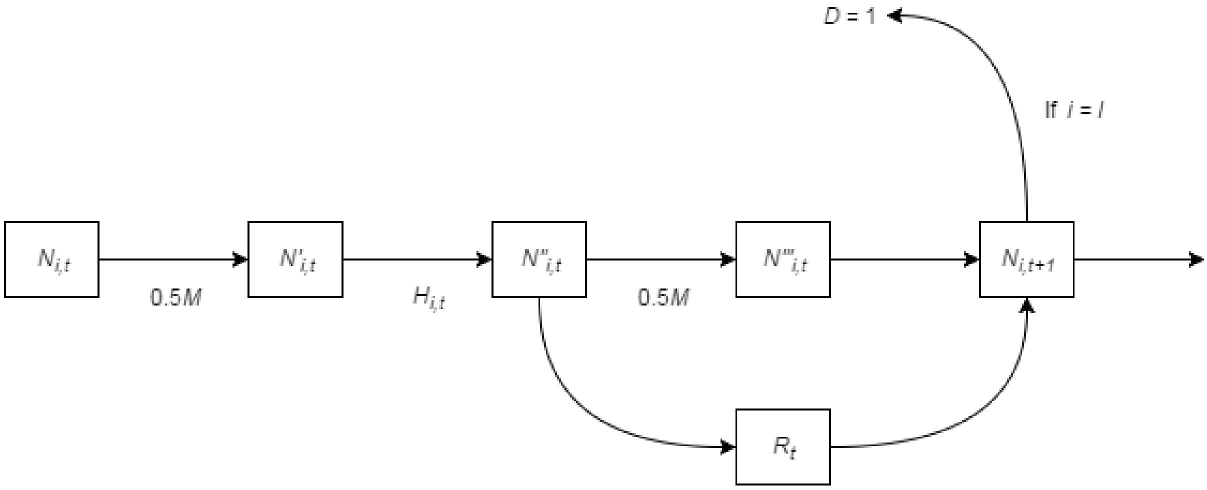


Figure 4.6: Conceptual representation of the hilsa population model. Subscripts i and t represent age class and year. Parameters M, H and R represent the instantaneous rate of natural mortality, number of fish harvested, and recruitment. In order to avoid making assumptions about when M is highest, I assumed that the first half of M took place before harvesting and recruitment and the second half after. Parameter D represents age class mortality. The final age class, I, was assumed to die each year (D = 1). Final arrow indicates that the process is repeated.

To calculate the number of fish surviving natural mortality, M, at age i and time t, $N'_{i,t,}$ I first applied half of M using the equation:

$$N'_{i,t} = N_{i,t}e^{-0.5M} \tag{3.1}$$

To then apply F without confounding total mortality with selectivity, I divided the total number of fish harvested at time t, H_t, proportionately across selected age classes, i, where N'_H is the total population of harvestable fish after instantaneous natural mortality, M, at time t, F is the instantaneous rate of fishing mortality and d is a dummy variable = 1 if the age class is selected, or = 0 if not:

$$N'_H = \Sigma dN'_{i,t} \tag{3.2}$$

$$H_t = N'_H(1 - e^{-F}) \tag{3.3}$$

$$H_{i,t} = H_t \left(\frac{N'_{i,t}}{N'}\right) d \tag{3.4}$$

$$N''_{i,t} = N'_{i,t} - H_{i,t} \tag{3.5}$$

I calculated recruitment at time t, R_t, using the following equations, where S_t is spawning stock size at time t, a and b are constants from the Beverton-Holt S-R relationship (Table 4.2), $r_{i,t}$ is proportion mature in age class i at time t, N' is the sum of N'_i, I is maximum number of age classes. Total S_t was multiplied by the proportion of females represented by the sex ratio (0.6), to account for only females spawning.

$$S_t = \left(\sum_{i=I}^{I} r_{i,t} N''\right) 0.6 \tag{3.6}$$

$$R_t = \frac{aS_t}{b + S_t} \tag{3.7}$$

Then I applied the second half of M:

$$N'''_{i,t} = N''_{i,t}e^{-0.5M} \tag{3.8}$$

After which all age classes were assumed to age one year, apart from the final age class which was assumed to die.

4.2.7 Calculating the ecological impact index

With the YPR analysis as justification, I ran the population model with F = 0.8 under non-selective fishing and under two selectivity regimes: one that just targets *jatka* and one that targets only adult fish. I used the equilibrium population size (t = 30 years) for each fishing regime to calculate the difference between regimes in the ecological impact on the population's productivity, E, relative to the regime with the lowest impact, using the following equation, where N_p is the least damaging and N_a is the alternative regime:

$$E = \frac{N_p - N_a}{N_p} \tag{4.1}$$

I tested the sensitivity of the analysis to the assumption of $F_{\max}$ = 0.8 by running the model with double (1.6) and half (0.4) $F_{\max}$, which gives an indication of the relative impact of these fishing activities under higher and lower levels of exploitation. I scaled the index to 1 for N_p.

When multiplied by reported hilsa catch volume (kg), these values gave an overall index of potential relative ecological impact for each household, dependent on the reported age of fish caught by that household and the household's reported average catch volume in peak and lean fishing seasons, in the context of the overall F assumed to pertain in the fishery as a whole.

4.2.8 Linear mixed effects modelling

Linear mixed effects models (LMMs) were used to model household threat to hilsa under $F_{\max}$, which was assumed to reflect reality in the overall fishery. This analysis was conducted in R version 3.2.3 (R Development Core Team 2016) with the R package lme4 (Bates et al. 2015). The response variable (potential relative household ecological impact) was log transformed for normalisation and reduction of heteroskedasticity. LMMs were fitted as random intercept models with district and village as grouping factors in the random effects, according to the sampling design (see Appendix B.1). In order to accurately estimate the corrected Akaike's Information Criterion (AICc) and Akaike weights for each model, all possible combinations of explanatory variables were fitted using Maximum Likelihood (ML) estimation procedures with the R package MuMIn (Barton 2011). The top candidate models were selected according to their AICc, and those with **Δ**AICc < 4 were re-run using Restricted Maximum Likelihood (REML) estimation procedures for accurate parameter estimates (Zuur et al. 2009). If candidate models had a **Δ**AICc value of less than two, the most parsimonious model was selected; otherwise the model with the lowest AICc was selected (Burnham & Andersen, 2002). Final models were checked for residual normality, heteroskedasticity and correlations between fixed effects and the residuals. The best random effects structures were selected using likelihood ratio tests and validation plots (Bolker et al. 2009). To analyse spatial effects, I estimated best linear unbiased predictors (BLUPs) from the top models, which measured the residual effect associated with each random effect (district and village within district). Marginal and conditional R^2 statistics were calculated following Nakagawa & Schielzeth (2013); the marginal R^2 represents the variance explained by the fixed effects, whereas the conditional R^2 represents that explained by the whole model.

Fishing location, fishing dependence, boat ownership[16], gear diversity, sanctuary area and compensation area were investigated as fixed effects (Table 4.3) in three separate global models (Table 4.4). Fishing location, boat ownership and gear diversity were all expected to influence ecological impact through their potential effects on selectivity and catch volume. Sanctuary area and compensation area were also expected to influence ecological impact because the availability of hilsa and *jatka* may vary between these environments. Similarly, ecological impact was expected to vary spatially between village and district. Receiving compensation would be expected to influence ecological impact if the compensation were effective, but if compensation is targeted towards *jatka* fishers and it is not effective in reducing *jatka* fishing, then a positive influence on ecological impact might be expected. Fishing dependence was represented by an index developed in Chapter 6, which primarily contrasts households with a

[16] It is common for fishers to fish in groups on one boat owned by a relatively wealthy individual who does not fish but retains a large proportion of the catch or profit.

high dependence on fishing (those who own boats, use multiple fishing gears, fish illegally, have higher proportions of income from fishing, no agricultural land or other livelihoods) with less dependent households (those who have agricultural land and other livelihoods, do not own boats, use a single gear type, do not fish illegally and have lower proportions of income from fishing). Fishing dependence might be expected to have a positive effect on ecological impact because *jatka* fishers, who are generally viewed to have the greatest impact on hilsa populations, also tend to be more dependent on fishing than non-*jatka* fishers, according to results in Chapter 6. However, dependence is not necessarily positively associated with catch volume. In fact, households who are highly dependent may not have the means to catch large volumes of hilsa, and households who do catch large volumes are not necessarily highly dependent (Marshall et al. 2007). Collinearity among explanatory variables was explored using pairwise plots, chi-squared tests, and phi coefficients. Although there were some significant associations (see Section 4.3.1), correlations were weak ($-0.5 > \phi < 0.5$, $p < 0.05$) or the variables were deemed to be different enough to have independent meaning. No meaningful interactions were found between explanatory variables. Exclusions due to missing data left sample sizes of 669 for Models 1 and 3, and 738 for Model 2.

Table 4.3: List, type and description of variables investigated through LMMs for potential relative ecological impact.

Variables	Type	Description	Expected influence	Rationale
Fixed effects				
Boat ownership	Binary	Households owns a boat (1) or does not own a boat (0)	+	Households who own boats are likely to have the capacity to catch a greater volume of hilsa.
Fishing location	Binary	Fishes in sea and river (1) or fishes only in river (0)	+/-	Catch characteristics may differ between those who fish in the sea and those who do not, due to hilsa life cycle, as may the impact of management (which is primarily targeting inland areas).
Sanctuary area	Binary	Household lives within a sanctuary (1) or outside a sanctuary (0)	+/-	Households living in sanctuary areas may catch larger volumes of hilsa and target *jatka* more because these areas are thought to be most important for hilsa and may have benefited from management. On the other hand, targeting *jatka* may be less likely in these areas because enforcement efforts are focused here.
Compensation area	Binary	Household lives in area where the compensation scheme operates (1) or in a control area (0)	+/-	Households outside of the compensation area may catch larger volumes of hilsa and target *jatka* more than those inside because there is less management in these areas. On the other hand, there may be lower availability of hilsa and *jatka* in these areas.
Compensation recipient	Binary	Household receives compensation (1) or not (0)	+/-	Compensation is officially targeted towards *jatka* fishers, but recipients should be less likely to target *jatka* if the compensation is having its intended impact
Gear diversity	Binary	Household uses 1 gear type (1) or > 1 gear type (0)	+/-	Households who focus on one gear type may target *jatka*, but those that use many gear types may catch more fish
Fishing dependence	Continuous	Index developed in Chapter 6	+/-	Households who are very dependent on fishing tend to target *jatka*, but they also tend to be poor and are unlikely to have the capacity to catch the greatest volume of hilsa.
Random effects				
District	Categorical	6 level factor (models 1 & 3); or 8 level factor (model 2)		
Village	Categorical	19 level factor (models 1 & 3); or 23 level factor (model 2)		

Table 4.4: Fixed effects included in each LMM for potential relative ecological impact of fishing households. Compensation area and sanctuary area were included in separate models for ease of interpretation: it is not possible for a household living outside of an area covered by the compensation scheme to also be living inside a sanctuary. Fishing dependence was included in a separate model because no values for fishing dependence were available for households outside of the compensation area, and because it incorporates boat ownership and gear diversity.

Model 1	**Model 2**	**Model 3**
Sanctuary area	Compensation area	Fishing dependence
Compensation recipient	Boat ownership	
Boat ownership	Fishing location	
Fishing location	Gear diversity	
Gear diversity		

4.3 Results

4.3.1 Characterising the fishery

Forty-seven per cent of hilsa fishing households in the study area reported use of only one gear type, 36 per cent reported use of two gear types and 15 per cent reported use of three gear types. In total, 22 different gear types were reported, but very few households reported use of more than three types (2 per cent). The most commonly reported gear was large-mesh *chandi jal*, a gillnet used mainly to catch hilsa (65 per cent of households); followed by *current jal*, an illegal monofilament gillnet used to target *jatka* (37 per cent of households); *behundi jal*, a set bagnet that targets a wide range of juvenile fish and small species that cannot move against the current (16 per cent of households); and *poa jal*, a gillnet that targets hilsa and some other fish (15 per cent of households). Fifty-four per cent of respondents only reported catching hilsa and/or *jatka,* while the remainder reported targeting other species as well. For self-identified *jatka* fishers, the most commonly reported gear was *current jal* (70 per cent of households), followed by *chandi jal* (50 per cent of households). Sixty-six per cent of households reported boat ownership, and 33 per cent reported fishing in both rivers and the sea, as opposed to the 67 per cent who reported fishing only in rivers.

Significant associations were found between some household characteristics and fishing activities (see Table 4.5). For instance, living inside the compensation area was positively associated with river fishing (as opposed to sea fishing), with using more than one gear type, and with boat ownership. Living inside a sanctuary area was also positively associated with river fishing, and with more diverse gear. More diverse gear was positively associated with boat ownership and sea fishing, and sea fishing and boat ownership were also positively associated. Sea fishers reported significantly larger catches in peak and lean seasons. In peak season, households owning boats reported significantly larger catches. Reported catch was significantly higher by households outside the compensation area than inside, and inside sanctuary areas than outside, in both seasons. As would be expected from the hilsa life cycle, *jatka* fishing was positively associated with river fishing (Fig. 4.7). *Jatka* fishing was also positively associated with diverse gear (Fig. 4.8) and boat ownership.

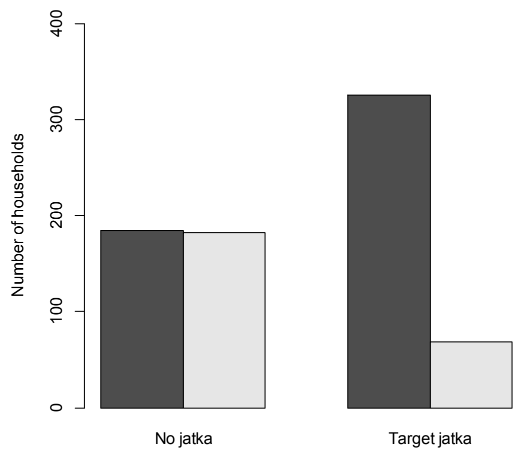


Figure 4.7: Bar plot showing the association between fishing location (dark grey = rivers, light grey = sea and rivers) and *jatka* fishing.

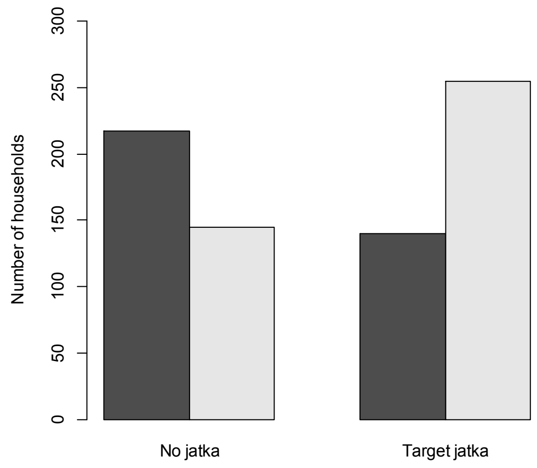


Figure 4.8: Bar plot showing the relationship between gear diversity (dark grey = 1 gear type, light grey = >1 gear type) and targeting *jatka*.

Table 4.5: Summary statistics of household characteristics and results of significant associations between them.

Variables	**Summary statistics**	**Significant associations**	
Sanctuary area (1 = inside, 0 = outside)	1 = 65%	Gear diversity (+)	$\chi^2 = 6.15$, df = 1, $p < 0.05$
		Sea fishing (-)	$\chi^2 = 12.17$; $p < 0.001$
		Peak season catch volume (+)	Wilcoxon rank sum test W = 49418, $p < 0.0001$
		Lean season catch volume (+)	Wilcoxon rank sum test W = 46146, $p < 0.0001$
Compensation area (1 = inside, 0 = outside)	1 = 85%	Gear diversity (+)	$\chi^2 = 11.27$, df = 1, $p < 0.0001$
		Boat ownership (+)	$\chi^2 = 38.37$, df = 1, $p < 0.0001$
		Sea fishing (-)	$\chi^2 = 13.28$, df = 1, $p < 0.001$
		Peak season catch volume (-)	Wilcoxon rank sum test W = 13875, $p < 0.0001$
		Lean season catch volume (-)	Wilcoxon rank sum test W = 11444, $p < 0.0001$
Compensation recipient (1 = recipient, 0 = non-recipient)	1 = 48%	Sea fishing (-)	$\chi^2 = 8.03$, df = 1, $p < 0.01$
		Boat ownership (+)	$\chi^2 = 6.79$, df = 1, $p < 0.01$
		See also Chapter 6	
Gear diversity (1 = > 1 gear type, 0 = 1 gear type)	1 = 52%	Boat ownership (+)	$\chi^2 = 7.32$, df = 1, $p < 0.01$
		Jatka fishing (+)	$\chi^2 = 44.53$, df = 1, $p < 0.0001$
		Sanctuary area (+)	$\chi^2 = 6.15$, df = 1, $p < 0.05$
		Compensation area (+)	$\chi^2 = 11.27$, df = 1, $p < 0.0001$
Boat ownership (1 = boat, 0 = no boat)	1 = 66%	Sea fishing (+)	$\chi^2 = 6.15$, df = 1, $p < 0.05$
		Jatka fishing (+)	$\chi^2 = 53.87$, df = 1, $p < 0.0001$
		Peak season catch volume (+)	Wilcoxon rank sum test W = 51257, $p < 0.001$
		Compensation area (+)	$\chi^2 = 38.37$, df = 1, $p < 0.0001$
		Compensation recipient (+)	$\chi^2 = 6.79$, df = 1, $p < 0.01$
		Gear diversity (+)	$\chi^2 = 7.32$, df = 1, $p < 0.01$
Fishing location (1 = sea and river, 0 = river only)	1 = 33%	*Jatka* fishing (-)	$\chi^2 = 87.89$, df = 1, $p < 0.001$
		Peak season catch volume (+)	Wilcoxon rank sum test W = 52387, $p < 0.01$

		Lean season catch volume (+)	Wilcoxon rank sum test W = 51681, $p < 0.01$
		Sanctuary area (-)	$\chi^2 = 12.17$; $p < 0.001$
		Compensation area (-)	$\chi^2 = 13.28$, df = 1, $p < 0.001$
		Compensation recipient (-)	$\chi^2 = 8.03$, df = 1, $p < 0.01$
		Boat ownership (+)	$\chi^2 = 6.15$, df = 1, $p < 0.05$
Jatka fishing (1 = yes, 0 = no)	1 = 52%	Boat ownership (+)	$\chi^2 = 53.87$, df = 1, $p < 0.0001$
		Sea fishing (-)	$\chi^2 = 87.89$, df = 1, $p < 0.001$
		Gear diversity (+)	$\chi^2 = 44.53$, df = 1, $p < 0.0001$
Average hilsa catch volume per fishing trip in peak season (kg)	Mean = 2.33; median = 2.00; SE = 0.06	Sanctuary area (+)	Wilcoxon rank sum test W = 49418, $p < 0.0001$
		Compensation area (+)	Wilcoxon rank sum test W = 46146, $p < 0.0001$
		Sea fishing (+)	Wilcoxon rank sum test W = 52387, $p < 0.01$
		Boat ownership (+)	Wilcoxon rank sum test W = 51257, $p < 0.001$
Average hilsa catch volume per fishing trip in lean season (kg)	Mean = 0.90; median = 0.50; SE = 0.03	Sanctuary area (+)	Wilcoxon rank sum test W = 46146, $p < 0.0001$
		Compensation area (+)	Wilcoxon rank sum test W = 11444, $p < 0.0001$
		Sea fishing (+)	Wilcoxon rank sum test W = 51681, $p < 0.01$
Fishing dependence	Mean = 0.17; median = 0.10; SE = 0.02	See Chapter 6	

4.3.2 Ecological impact

The potential relative impact of each selectivity regime on hilsa biomass under F_{max} ranged from 1 to 1.177 (Table 4.6). Targeting *jatka* was consistently the least damaging selectivity regime, under each level of F, and fishing across all age classes was consistently the most damaging, with the relative differences ranging from 6.1 per cent at the lowest F to 24.7 per cent at the highest F.

Table 4.6: Relative levels of impact on hilsa population equilibrium biomass for each selectivity regime, and equilibrium biomass under the least damaging regime (targeting *jatka*) as a percentage of virgin biomass, under low (F_{max} x 0.5), medium (F_{max}) and high (F_{max} x 2) levels of F, where F is the instantaneous rate of fishing mortality, and F_{max} is the F at which maximum YPR was achieved.

Fishing mortality	**Selectivity regime impact**			**Biomass under least damaging regime as percentage of virgin biomass**
	Targeting *jatka*	**Targeting adults**	**All age classes**	
F_{max} x 0.5 (0.4)	1.000	1.061	1.111	94
F_{max} (0.8)	1.000	1.094	1.177	90
F_{max} x 2 (1.6)	1.000	1.119	1.247	83

The range of the household-level ecological impact index was much larger (0.5-16.4 in peak season and 0-5.5 in lean season; Fig. 4.9). There was a strong positive correlation in household-level ecological impact between fishing seasons, reflecting overall exploitation regimes in each season under F_{max} (Pearson's product moment correlation coefficient = 0.92, $p < 0.0001$). Because much greater volumes of fish are caught in peak season, subsequent analyses are presented only for peak season.

The index was dominated much more by catch volume than selectivity. Because the *jatka* fishing variable was binary, only two of the selectivity regimes were represented in the household-level index (targeting adults and targeting *jatka*), and so the relative difference in selectivity under

F_{max} was only 9.4 per cent. In contrast, the range of reported hilsa catch volumes was much wider (Fig. 4.10).

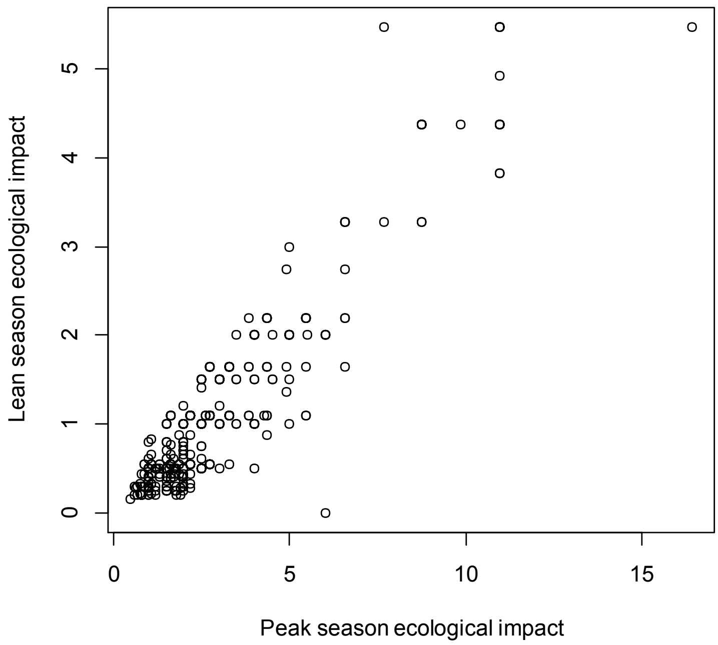


Figure 4.9: Correlation of household-level potential relative ecological impact between peak and lean fishing seasons.

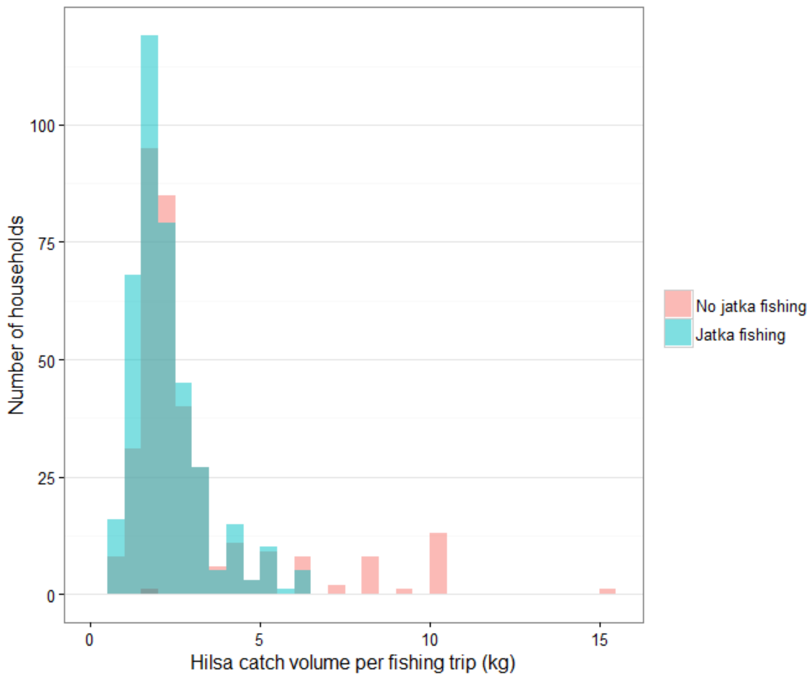


Figure 4.10: Overlaid histogram showing the range of average volumes of hilsa caught per fishing trip (kg) in peak season, as reported by fishing households who self-identified as jatka fishers and those who did not. The darker blue is where the two groups overlap.

4.3.3 Correlates of household-level ecological impact threat

Boat ownership and fishing location were included as fixed effects in the final selected model for Model 1 (Table 4.7). The linear coefficients estimated by the model indicated that the households with the greatest potential ecological impact are those who own boats and fish in the sea. Living in sanctuary areas, gear diversity and receiving compensation were not significant explanatory variables (See Table B.2 for model selection). The top models (with and without sanctuary area) had very similar AICc values, but the sanctuary area had a high standard error, and so was not informative. The most parsimonious model was therefore selected.

Plotting the BLUPs for each district (Fig. 4.11a) also showed an effect of geography on ecological impact, once fixed effects were taken into account. They indicated that households in Chandpur district have the greatest potential ecological impact, while those in Barisal have the lowest. An effect can also be seen of village on potential ecological impact (Fig. B.3a), but such fine-scale variation is beyond the scope of the study.

When sanctuary area and compensation recipient were removed from the global model to look instead at the effect of compensation area (Model 2), the effects of boat ownership and sea fishing remained similar, but the coefficient estimated for compensation area indicated that households living inside a compensation area have a lower potential ecological impact than those in control areas (Table 4.7; see Table B.3 for model selection). The BLUPs showed a similar pattern in terms of the district effect on ecological impact, and the most extreme districts were those outside the compensation areas (Fig. 4.11b). Households in Cox's Bazar district now appeared to have the greatest ecological impact, and those in Rajbari district the lowest (Fig. 4.12a). The conditional R^2 was much higher than the marginal R^2 for all models, indicating that the random effects explain the most substantial portion of variance.

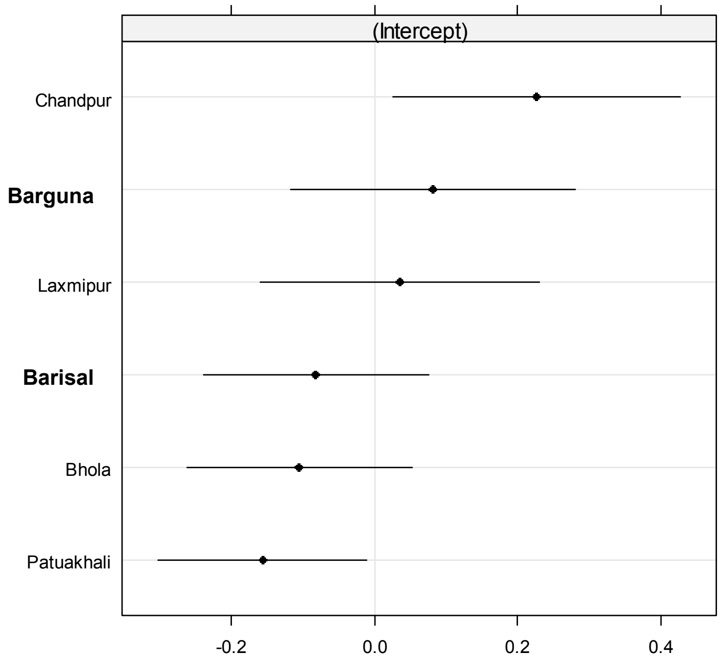
When fishing dependence was tested as a fixed effect, it was not a significant explanatory variable, and so it was not included in the final model for Model 3 (Table 4.7; see Table B.4 for model selection). BLUPs are not presented for Model 3 due to the lack of fixed effects.

Comparing the distribution of the BLUPs for the random effect of district on overall ecological impact (Fig. 4.12a) with those on the individual elements of selectivity (Fig. 4.12b) and catch volume (Fig. 4.12c), clearly demonstrates that the index of overall ecological impact is dominated more by catch volume than selectivity.

Table 4.7: Coefficient estimates (with standard errors) for LMMs with household-level ecological impact index as the response variable, where **(Model 1)** sanctuary and compensation recipient were included as fixed effects in the global model instead of compensation area; **(Model 2)** compensation area was included as a fixed effect instead of sanctuary and compensation recipient; and **(Model 3)** where only an index of fishing dependence was included rather than individual variables which make up the index (includes boat ownership). Random effects estimates of variance are also presented [with standard deviation]. Model1: R^2_m = 0.149; R^2_c = 0.510; based on 669 households from 19 villages in 6 districts. Model 2: R^2_m = 0.163; R^2_c = 0.735; based on 738 households from 23 villages in 8 districts. Model3: R^2_m = 0; R^2_c = 0.280; based on 669 households from 19 villages in 6 districts.

Model 1: Sanctuary and compensation recipient			**Model 2: Compensation area**			**Model 3: Fishing dependence**		
Fixed effects	***Estimate***	***t value***	***Fixed effects***	***Estimate***	***t value***	***Fixed effects***	***Estimate***	***t value***
Intercept	0.336 (0.125)	2.693	Intercept	0.342 (0.215)	1.588	Intercept	0.612	6.711
Boat ownership	0.381 (0.036)	10.672	Compensation area	- 0.778 (0.431)	1.806			
Fishing location	0.243 (0.055)	4.397	Boat ownership	0.373 (0.033)	11.156			
			Fishing location	0.254 (0.055)	4.652			
Random effects			***Random effects***			***Random effects***		
Village	0.026 [0.160]		Village	0.022 [0.147]		Village	0.101 [0.101]	
District	0.034 [0.185]		District	0.264 [0.514]		District	0.055 [0.234]	

(a)



(b)

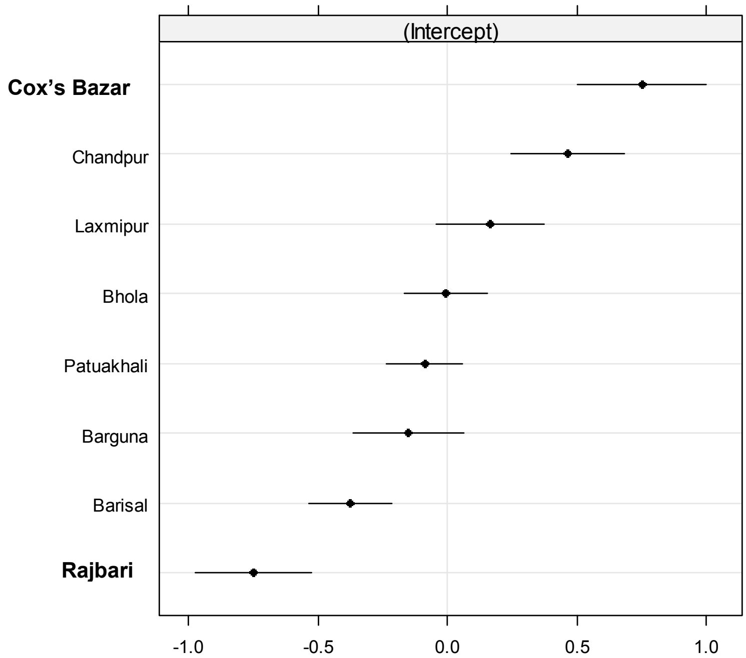


Figure 4.11: BLUPs for the random effect of district for **(a)** Model 1, where bold names represent districts that are outside sanctuary areas; and **(b)** Model 2, where bold names represent control areas outside compensation areas. The x axes show the effect of living in a particular district in terms of the difference in ecological impact from the intercept. Error bars show the 95% confidence interval based on the conditional variance for each random effect.

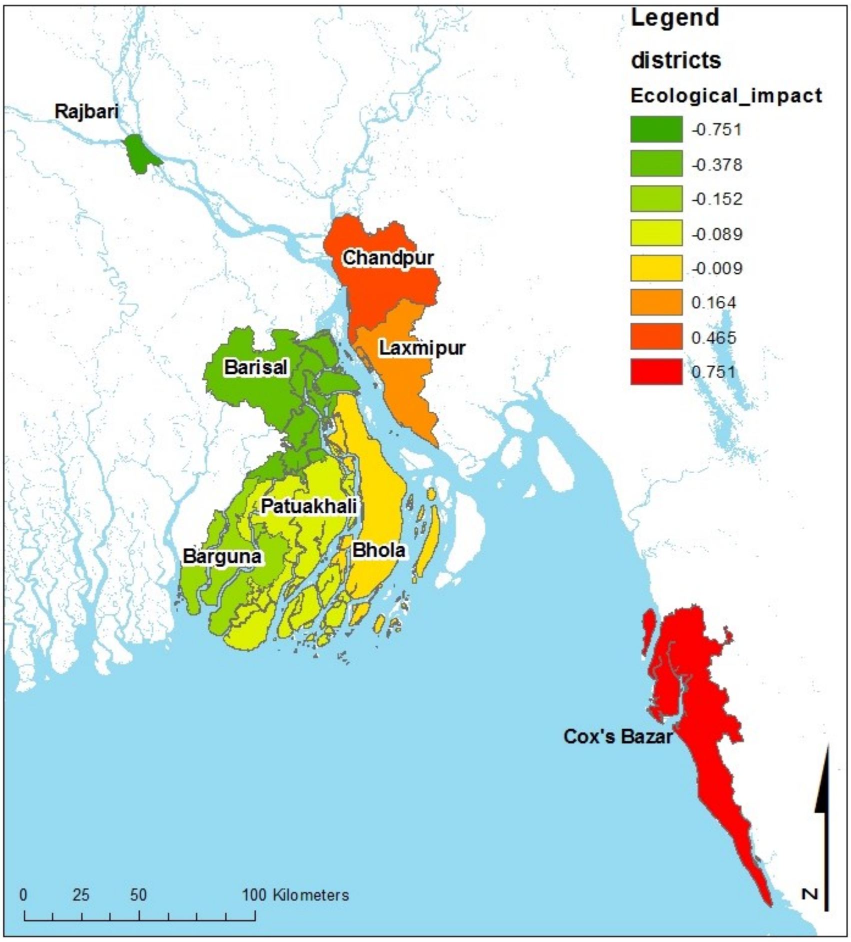


Figure 4.12a: Map showing the distribution of household-level ecological impact between districts, where -0.75 is low impact and +0.75 is high impact. Distribution derived from the BLUPS of Model 2.

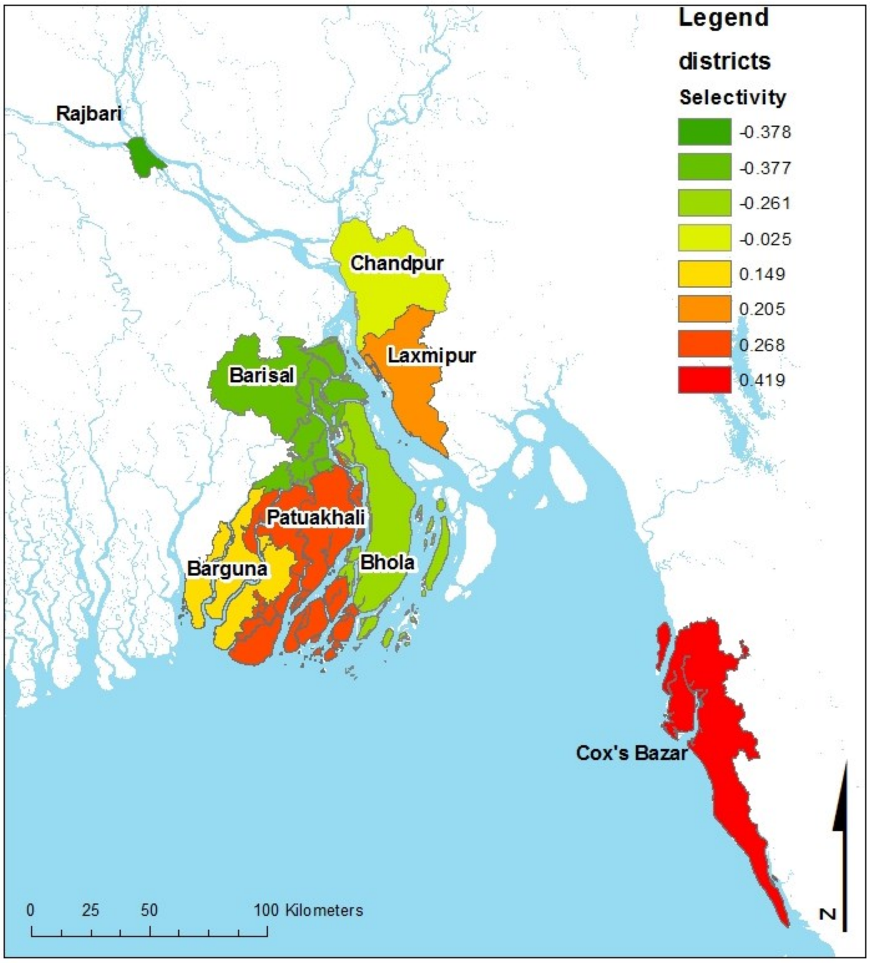


Figure 4.12b: Map showing the distribution of household-level impact (according to size selectivity only) between districts, where -0.038 is low impact (more *jatka* fishers) and 0.419 is high impact (fewer *jatka* fishers). The values were derived from a binomial generalised linear mixed effects model with the probability of targeting *jatka* as the response variable and district and village as grouping factors in the random effects, from which I extracted the best linear unbiased predictors for the district random effect (Table B.5).

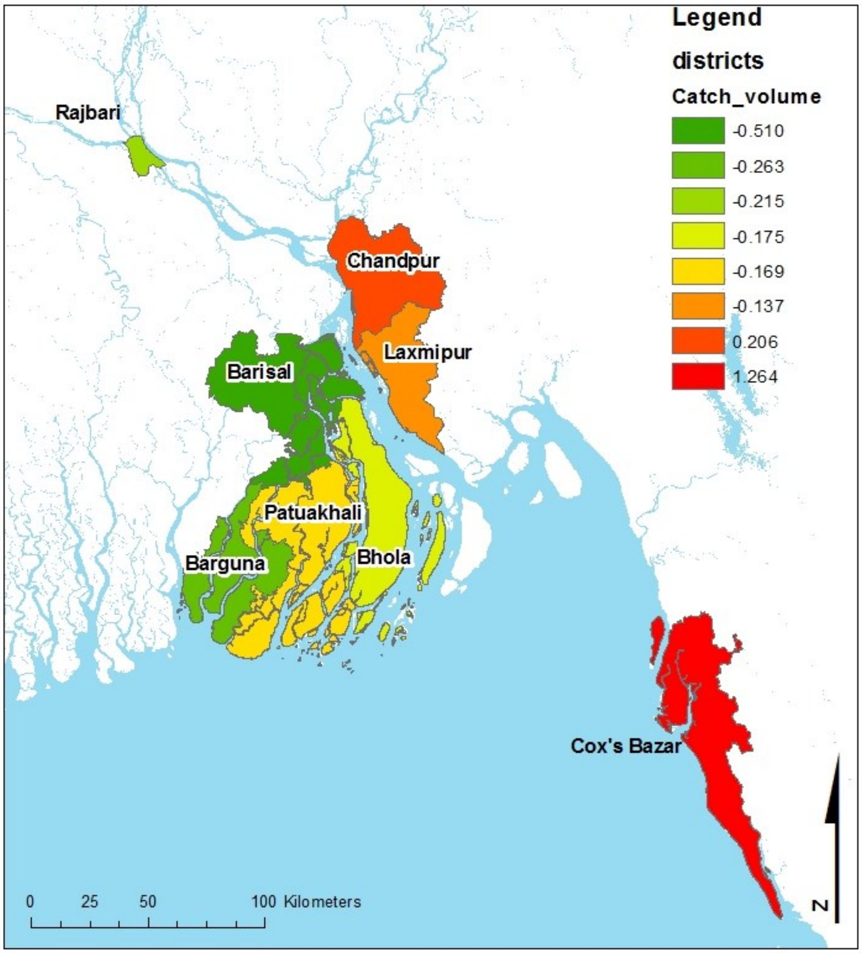


Figure 4.12c: Map showing the distribution of household-level impact (according to catch volume only) between districts, where -0.510 is low impact (smaller catch volumes) and 1.264 is high impact (greater catch volumes). The values were derived from a linear mixed effects model with catch volume (log transformed) as the response variable and district and village as grouping variables in the random effects, from which I extracted the best linear unbiased predictors for the district random effect (Table B.5).

4.4 Discussion

4.4.1 Fishery-level ecological impact

The relative impacts of harvesting *jatka* vs. adults on hilsa population biomass suggest that the current focus of hilsa management on the protection of *jatka* may not be the most effective approach. Under F_{max}, the impact of fishing adults was 9.4 per cent higher than that of fishing *jatka*, and the impact of fishing across all age classes was 17.7 per cent higher. The relatively low impact of fishing *jatka* can be explained by the steep S-R curve of hilsa, which is indicative of recruitment with very low density-dependence (Beddington & Kirkwood 2005). Under double F_{max}, the relative impacts of fishing adults and of fishing all age classes increased to 11.9 per cent and 24.9 per cent, respectively, and under half F_{max}, they decreased to 6.1 per cent and 11.1 per cent. This pattern indicates that hilsa populations are more sensitive to the rate of their exploitation than to size selectivity. Meta-analyses of large empirical datasets for North East Atlantic fish stocks have shown the effect of exploitation rates on stock status to be stronger than that of size selectivity (Vasilakopoulos et al. 2011, 2012), while slightly different methods have shown selectivity to be more influential for spawning stock biomass and yield (Vasilakopoulos et al. 2015). Ultimately, the relative impacts of exploitation rate and size selectivity will depend on F_{max}/M for a species, which tends to increase with recruitment steepness (Beddington & Kirkwood 2005; Zhou et al. 2012). For abundant, short-lived species like hilsa, where lots of juveniles are dying anyway, exploitation rate is expected to be more important.

An interaction can be seen between exploitation rate and size selectivity in hilsa: the doubling of F exacerbates the size of effect of size selectivity on hilsa biomass, although the pattern of the effect remains fairly constant. The effect might have been expected to get much larger at high F, but even at double F_{max} targeting *jatka* only reduced equilibrium biomass to 83 per cent of virgin biomass. This suggests that, as a result of early maturation, high intrinsic growth rate and high reproductive rates, hilsa are very resilient to high levels of F. These results are consistent with studies that show evidence of high resilience in clupeids (Hutchings 2000; Thorpe et al. 2015) and of higher sensitivity to exploitation, and thus lower fishing mortality reference points, in larger species of fish (Le Quesne & Jennings 2012; Fung et al. 2015).

4.4.2 Household-level ecological impact

Household-level relative potential ecological impact, as defined by reported hilsa catch volume and size selectivity, follows a spatial pattern clearly reflected in the random effect of district. Modelling indicated that households in Cox's Bazar have the greatest ecological impact, followed by those in Chandpur, whereas those in Rajbari, Barisal, and Barguna have the lowest impact. Breaking the index of ecological impact down into its individual components of selectivity and catch volume showed that these district-level differences were driven largely by catch volume. Households in Cox's Bazar and Chandpur reported the greatest hilsa catch volumes, and Barisal, Barguna and Rajbari the least. Cox's Bazar is a district with both private and Government-owned marine landings centres, and Chandpur is a major inland hilsa landings site, so it follows that fishing effort is high in these districts where households have easy access to stocks. The distribution of size selectivity was quite different (more *jatka* fishing households in Rajbari, Barisal, and Bhola than in Cox's Bazar, Patuakhali, and Laxmipur), but this was not reflected so much in the index because selectivity only puts a 9.4 per cent weighting on catch volume.

Modelling also showed that households who own boats and fish in the sea have a greater potential ecological impact than those without boats and fishing only in rivers, although these factors were much less important than spatial factors. Given the positive associations of boat ownership and sea fishing with greater catch volumes, and the clear dominance of catch volume in the impact index, it follows that these effects are largely driven by their influence on catch volumes. This could be a reflection of the state of stocks in marine vs. riverine environments, or it could be a reflection of levels of fishing effort: coastal fishers are more likely to use larger mechanised boats and therefore have a greater fishing capacity. River fishers, on the other hand, are more likely to be *jatka* fishers, and so have less influence on the index. Although boat ownership was also significantly associated with more diverse gear use (as was *jatka* fishing) there was no significant association between gear diversity and catch volume, and modelling did not show a significant effect of gear diversity on ecological impact.

Living inside a compensation area was a significant negative correlate of ecological impact. This can be explained by the two extremes represented by the control districts. Cox's Bazar is characterised by large hilsa catch volumes and few *jatka* fishers, whereas Rajbari is characterised by low hilsa catch volumes but lots of *jatka* fishing. This could be interpreted as evidence of an inappropriate spatial focus of the compensation scheme, not just socially (Chapter 6) but also in ecological terms (Chapter 5). On the other hand, households living in areas where the compensation scheme operates may be less likely to target *jatka* as a result of the scheme and associated management, as is intended – but there is limited evidence for this (Chapter 5). Although the existence of a sanctuary area was not found to be a significant correlate in this study, it would be worth further investigation. Management is more intensive inside sanctuary areas, which could be having a positive impact on catch volume. On the other hand, this impact could be masked by the fact that any compliance with the sanctuary fishing bans should actually reduce catch volumes during peak season.

The fact that fishing dependence was not a significant correlate of ecological impact indicates that the households most in need of compensation, in terms of vulnerability reduction (see Chapter 6), are not the same ones who are catching the most hilsa, and hence who have the greatest potential to contribute to ecological additionality. This is the same result that would be expected if households had already reduced their ecological impact as a result of compensation, but receiving compensation had no significant effect on ecological impact and evidence from Chapter 5 suggests that such an effect is unlikely.

4.4.3 Study limitations

The simplicity of the population model used in this study imposed some limitations: for example, it assumed that all selected age classes are equally vulnerable to capture when, in fact, different gears have different selection patterns; it assumed constant M across all age classes; and it assumed a jump from 0 to 100 per cent maturity from the first age class to the second, which is not realistic. As a deterministic model it also assumed constant recruitment, ignoring stochastic fluctuations, but deterministic analyses can provide reasonable guides to average behaviour of stocks (Beddington & Kirkwood 2005). Moreover, the model did not take into account that large, old hilsa may have a much higher fecundity than smaller and younger fish. If the data were available, further investigation might show more severe biomass impacts of targeting adult fish, potentially in specific spawning areas. For analysis of impact at the household level, measures to improve the quality of the household survey data on size selectivity would therefore be useful, allowing an exploration of the impact on biomass of catching small, medium and large fish (rather than just *jatka* vs. adults) and thus of the potential impact of selecting or protecting megaspawners, which could be just as, or more, important (Froese 2004). Ideally, further investigation should be based on actual fishing behaviour, rather than reported average selectivity and catch volumes, which can be unreliable. More precision on where, exactly, households fish, beyond the district in which they live and whether they fish in the sea or not, might also shed more light on the spatial distribution of household-level impact.

It is important to acknowledge the potential uncertainties in the LHP values used in this study – particularly steepness, which is much less easy to estimate than other parameters and was not specifically estimated for hilsa (Beddington & Kirkwood 2005; Zhou et al. 2012). Within the Clupeidae family, estimates of steepness range from 2.1 to 31.9, and it is not known how close the median value actually is to the specific value for hilsa (Myers et al. 1999). Furthermore, the analysis of household-level impact depends on the assumption of $F = 0.8$, but predicting reference points – whether from established relationships among life history traits or directly from life history parameter (LHP) data – entails uncertainties (Le Quesne & Jennings 2012; Garcia-Carreras 2015; Thorpe et al. 2015). Per-recruit models are very sensitive to changes in input parameters, and the value of F_{max} in particular, as opposed to, say, $F_{0.1}$, is sensitive to small changes in these parameters (Hordyk et al. 2014, 2015; Garcia-Carreras 2015). However, the sensitivity analysis showed final results to be minimally affected by doubling and halving *F.* It is unclear to what extent these rates actually reflect the reality of the hilsa fishery, but given that estimates of F for hilsa range from 1.25 up to 2.49 (Chapter 3), it is probable that reference points used in this study are reasonably conservative.

Finally, the analysis of overall fishery-level impact assumed that all households follow the same fishing regime (i.e., fishing *jatka* or adults) under a reference F, while the analysis of household-level impact applied that overall selectivity impact, relative to the situation in which everyone is following the least damaging fishing activity, in a context where a wide range of selectivity and catch volume combinations actually exist. This is of course an oversimplification, but a useful first step to understanding the hilsa fishery in the absence of a proper stock assessment. The study adds to a growing body of literature showing that rapid assessments and prediction of biological reference points are possible even when only minimal LHP data are available, and that some form of management advice can still be provided in such circumstances (Beddington & Kirkwood 2005; Zhou et al. 2011; Le Quesne & Jennings 2012; Garcia-Carreras 2015; Thorpe et al. 2015).

4.4.4 Conclusions

The implications of this research for fisheries policy in Bangladesh are two-fold: firstly, it does not support the current focus of management on the protection of *jatka*; and, secondly, it provides evidence of the potential benefits of effort reduction. Calls have been made for an increased focus on input controls such as entry regulations, as opposed to output controls such as size limits, in small-scale, developing-world fisheries (Purcell & Pomeroy 2015). Recommendations for increased effort control in the hilsa fishery have been made before and rejected on the basis that it would not be practical when monitoring and enforcement are so poor, and that it would simply result in increased competition between vessels (Mome & Arnason 2007). However, the potential value – both in terms of hilsa biomass and the socioeconomic wellbeing of hilsa fishers – of focusing on moderating rates of exploitation should be considered.

It should also be noted that the impacts of a particular selectivity regime depend on what scale one is working at: the target population, the fish community or the entire ecosystem (Fauconett & Rochet 2016). As in most commercial fisheries, full implementation of balanced harvesting as part of EBFM may not be viable in Bangladesh, where human preferences and markets drive demand for hilsa over other species, and where the capacity and detailed knowledge for the approach have not yet been developed (Breen et al. 2016; Reid et al. 2016). However, given the weaknesses of single-species management (Chapter 5), development of this work to explore ecological impact more broadly, using multi-species models, would be valuable in steering management towards a more sustainable ecosystem-based approach.

Chapter 5

Exploring the ecological additionality of hilsa fishery management in Bangladesh

5.1 Introduction

Conservation interventions should have ecological additionality; i.e., measureable conservation benefit over and above what would have happened anyway (Maron et al. 2013). Uncertainties surrounding this additionality raise questions around cost-effectiveness (Wunder et al. 2008; Narloch et al. 2011), which, given the limited resources available, is essential in developing-world fisheries management. But, as discussed in Chapters 2 and 3, additionality is difficult to measure and commonly ignored. Impact evaluations in the conservation literature have started to incorporate counterfactual scenarios (e.g. Sánchez-Azofeifa et al. 2007; Andam et al. 2008; Pagiola & Rios 2008; Gurney et al. 2015), but it is only the use of *ex-ante* counterfactual scenarios which allows attribution of additionality to specific actions or behaviours, and since these are rarely developed early in the design process, attribution is often problematic (2013; Maron et al. 2013).

Nevertheless, when the data required by rigorous impact evaluations are unavailable, potentially useful studies of additionality can still be conducted by piecing together understanding from patchy and disparate primary data, and from a range of secondary sources. Although these approaches cannot provide assessments of true net benefit, they can provide an indication of confidence in or scope for additionality. For instance, in data-poor fisheries, extrapolation methods of evaluation such as the Robin Hood Approach (which steals information and understanding from data-rich fisheries to give to the data-poor) are sometimes the only tools available to managers, and can provide them with a starting point for the development of precautionary management methods (Honey et al. 2010). Understanding or data from similar species or neighbouring populations can also be used to help inform management decisions – an approach that has been explored in Australian multi-species trawl fisheries (Smith et al. 2009). Similarly, fisheries modelling studies often substitute data from similar species and stocks elsewhere when local data are unavailable (Pilling et al. 2008). For example, an exploration of marine mammal-fishery interactions in the Barents Sea transferred data from the North Sea, which is more data rich (Blanchard et al. 2002).

Expert or experiential knowledge and perceptions can also be useful when data are limited (Fazey et al. 2006; Bennett 2016). Fishers' ecological knowledge can complement the use of scientific knowledge in decision-making, and can be used to test hypotheses formulated using more conventional scientific knowledge (Neis et al. 1999; Johannes 2000; Wilson et al. 2006; Silvano & Valbo-Jørgensen 2008; Daw et al. 2011b; Gaspare et al. 2015). Because of their dependence on local resources, small-scale artisanal fishing communities can be highly aware of their impacts on the populations from which they harvest and provide valuable insights into their behaviour and ecology (Aswani & Hamilton 2004; Silvano & Begossi 2005, 2009; Drew 2005; Ramires et al. 2015). Stakeholder perceptions, which might be based on this knowledge, can support rapid impact assessments when environments for evaluation are challenging (Carey et al. 2003; Sainsbury et al. 2015). Often overlooked in the pursuit of scientific evidence of additionality, local perceptions are ultimately just as important in determining whether an intervention fails or succeeds (Bennett 2016).

The Bangladesh hilsa fishery is typical of the developing world in its limited availability and reliability of data. Through an analysis of the available literature, ecological data, local knowledge and perceptions, and understanding from other fisheries, this chapter will explore whether a package of carrot-and-stick management interventions in the Bangladesh hilsa fishery: a) has scope for additionality in the future; and b) has had any additional impact to date. I will first explore the potential for additionality through the development of a *post hoc* theory of change for hilsa management. Since hilsa management seeks to produce behavioural and ecological change (DoF 2002), an understanding of potential ecological impact must take into account not only ecological context but also institutional, economic and social contexts. Drawing on the field of evaluation (Rogers & Weiss 2007), theory of change is increasingly used in the design, implementation, and impact evaluation of initiatives intended to support development outcomes (Anderson 2005; Vogel 2012, 2013). The approach involves mapping out anticipated pathways between management interventions, the assumptions underlying these anticipated pathways, the issues and context they are seeking to influence, and the longer-term social, economic and ecological outcomes. As such, it is both a process and a product supporting critical analysis of the contextual conditions that influence the interventions and of the assumptions which underpin the logical pathways (Vogel 2013). Using the counterfactuals developed in Chapter 3 as a guide, I will explore the validity of my reconstructed theory of change by assessing the evidence behind the underpinning assumptions.

Next, I will explore recent spatial and temporal trends in the fishery and perceptions of current management, with a view to identifying any impacts which the management could potentially have had. Because data are not available for formal evaluation and perceptions are subjective, I cannot make a causal link between ecological trends (or perceived ecological trends) and management interventions, but correlative evidence can still be useful in fisheries management when experimental evidence is unavailable, as long as the full range of possible causal mechanisms are considered (Hilborn 2016). Even if additionality is not demonstrated in practice, I expect to see an indication of whether there is scope for current management to have had additionality. This should inform the development and improvement of hilsa fishery management interventions and provide an indication of whether an increased focus on either carrots or sticks would be more effective.

5.2 Methods

5.2.1 Data collection

Primary data was collected through key informant interviews (KIIs; i.e., interviews with individuals selected on the basis of their expert knowledge of the hilsa fishery) and a fishing household survey. I conducted unstructured interviews based around a list of key topics (e.g.

the constraints, limitations, and strengths of hilsa fisheries management, and the impact of the compensation scheme; see Appendix C.1) with 36 key informants (KIs) in Bangladesh. I selected KIs from local and international NGOs, Government officials from different tiers (central Government, local administrations, Department of Fisheries, and Bangladesh Fisheries Research Institute), academic institutions, and the hilsa supply chain in Chandpur – an important district for hilsa fishing and the one where the compensation scheme has been operating the longest (see Table 5.1). I placed emphasis on allowing the KIs to speak, only redirecting or encouraging them when they went off topic or had something particularly notable to say (Newing 2011). I conducted initial interviews in May 2014, but continued some conversations intermittently up until July 2015. The theory of change reconstruction was based on my understanding of current fishing regulations and associated management, as set out in the Hilsa Fisheries Management Action Plan (HFMAP; DoF 2002), on discussions with KIs, and on context from Chapter 3.

Table 5.1: Breakdown of key informants (KIs) by sector. Interviews are referred to in the text using codes.

Sector	Number of KIs	Code
Department of Fisheries (DoF) representatives[17]	6	D1-6
Local Government officials (non-DoF)	2	L1-2
Central Government officials[18] (non-DoF)	1	C1
Bangladesh Fisheries Research Institute (BFRI)* scientists	2	R1-2
Local NGO representatives	6	LN1-6
International NGO representatives	6	IN1-6
Industry stakeholders (middlemen)	5	M1-5
Industry stakeholders (members of fishers and boat owners associations)	3	F1-3
Academics working in Bangladesh	5	A1-5

*Autonomous organisation under administration of Ministry of Fisheries and Livestock

900 fishing households were interviewed in a survey described in detail in Appendix B.1. Households were sampled from within and outside sanctuary areas and from control sites outside the area where the compensation scheme operates (Fig. 5.1). Sections of the questionnaire used in this study asked about household characteristics, fishing activities, hilsa

[17] The DoF has representatives at each level of Government administration down to sub-district (see Chapter 3).

[18] See Chapter 3 and Appendix A.1 for a summary of the administrative hierarchy. District- and division-level administrations are led by central Government.

trends and status, and the compensation scheme (see Appendix C.2). The questionnaire did not address alternative livelihood support because coverage of this management approach is so low. The decision was made as a research team to use the last five years as a time frame, where questions required one, because there are no scientific data currently available for the fishery post-2009, and because little is known about the quality of information provided in interviews with long recall periods (Jones et al. 2008).

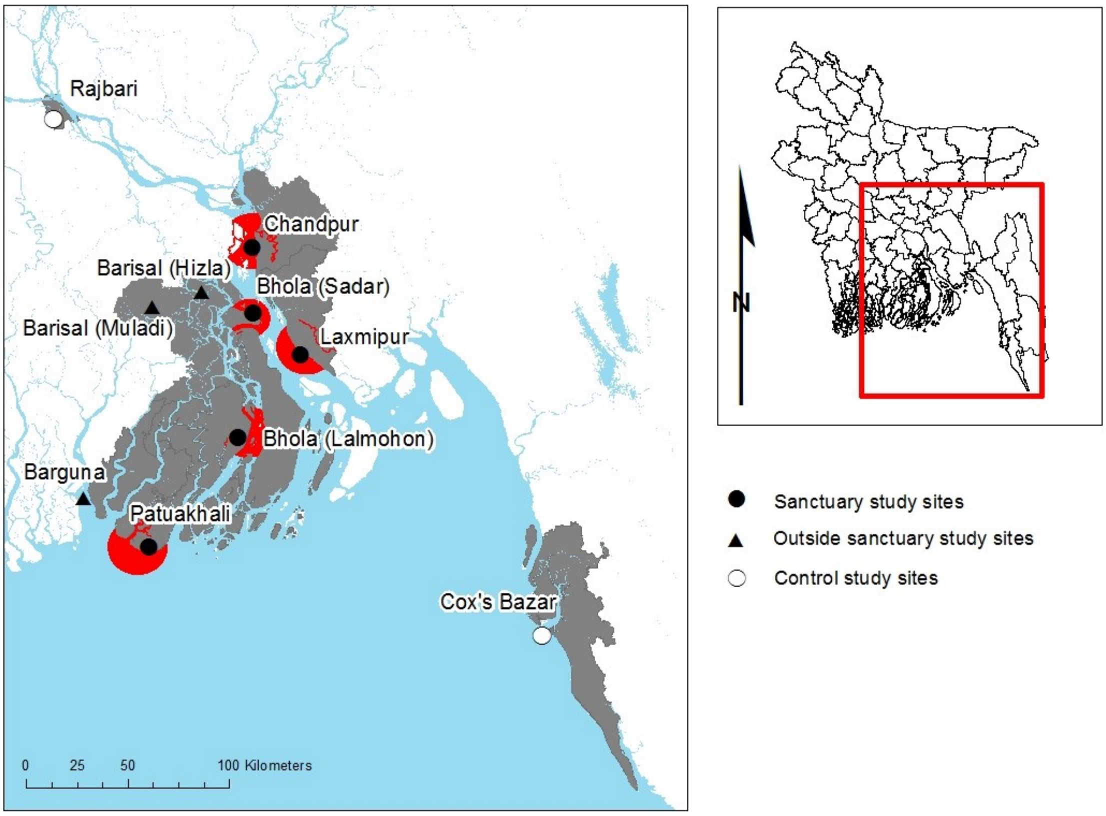


Figure 5.1: Map of study area, showing study site districts (grey) in relation to sanctuary sites (red). Control study sites are outside of the area where the compensation scheme operates. Each study site represents the approximate location of a cluster of surveyed villages, denoted by the relevant district name (precise village coordinates were not available). In Barisal and Bhola districts two village clusters were sampled and can be distinguished by the sub-district names (in brackets); in the other districts just one village cluster was sampled.

5.2.2 Statistical analysis

Where possible, binomial generalised linear mixed effects models (GLMMs) were used to test potential associations between perceptions and contextual variables. A GLMM allows data to exhibit correlation and non-constant variance, and so accommodates a range of types of response and explanatory variables, as well as random effects (Zuur et al. 2009). Models were fitted with: a) the probability of a fisher saying that hilsa catch was increasing from year to year; and b) the probability of a fisher saying that egg/fry presence was increasing from year to year as binary response variables (1 = increases; 0 = decreases or stays the same). There was not enough variation in answers to the question on fish size for statistical analysis. The GLMMs were fitted as random intercept models with district and village as grouping factors in the random effects and a logit link function. The best random effects structures were selected using likelihood ratio tests and validation plots (Bolker et al. 2009), and models were run with Laplace approximation using the package lme4 (Bates et al. 2015) in R version 3.2.3 (R Development Core Team 2016).

A summary and description of fixed effects can be found in Table 5.2. Continuous fixed effects were standardised by two standard deviations for direct comparison of coefficients following model averaging (Gelman 2008; Grueber et al. 2011). I included compensation area because if the compensation is having an impact on rule compliance or hilsa populations, or affecting answers, it might be expected that these effects would be seen more in areas covered by the scheme than control areas. I tested sanctuary area for a similar reason – if the fishing bans in the sanctuary areas are effective and having an impact on hilsa populations, the impacts are most likely to have been seen by households in these areas. The other fixed effects were included due to their potential influence on fisher perceptions, although the expected direction of influence was unclear for most of these.

Collinearity among explanatory variables was explored using pairwise plots, chi-squared tests, and phi coefficients. None of the variables were significantly correlated ($p > 0.05$) or only weakly correlated (-0.5 > ϕ < 0.5, p < 0.05). An information-theoretic approach to model selection was taken; all possible combinations of explanatory variables were fitted using Maximum Likelihood (ML) estimation procedures with the R package MuMIn (Barton 2016), and top candidate models were selected according to the corrected Akaike Information Criterion (AICc; Burnham & Anderson 2002; Bolker et al. 2009). No models were clearly superior (weights of top models were < 0.9), so those with **Δ**AICc < 4 were re-run using Restricted Maximum Likelihood (REML) estimation procedures for accurate parameter estimates (Zuur et al. 2009), which were then averaged across these models, allowing relative variable importance to be determined (Burnham & Anderson 2002; Grueber et al. 2011). Coefficients were presented for the full average, rather than the subset or conditional average, which has a tendency of biasing the values away from zero (Barton 2016). Models were checked for residual normality, heteroskedasticity and correlations between fixed effects and the residuals. To analyse spatial effects on the probability of receiving compensation, I estimated best linear unbiased predictors (BLUPs) from the global models, which measured the residual effect associated with each random effect (district and village within district). BLUPs, or conditional modes, can be conceptualised as the equivalent of the linear coefficients found for the explanatory variables (noting that they are not, strictly speaking, parameters). For all models, BLUPS were checked for bias in the order in which villages were visited and no pattern was found. Since women do not engage in hilsa fishing, 136 households with female respondents were excluded from analysis. 25 further households had missing data and were excluded from analysis.

Table 5.2: List, type and description of variables investigated through GLMMs for the probability of saying hilsa catch (kg) or egg/fry presence was increasing from year to year.

Variables	Type	Description	Expected influence	Rationale
Fixed effects				
Sanctuary area[19]	Binary	Household lives within a sanctuary (1) or outside a sanctuary (0)	+	If fishing bans are effective then households fishing inside or near sanctuaries should be more likely to perceive an increase in catch than those fishing outside.
Jatka[20] fishing	Binary	Household targets *jatka* (1) or not (0)	+/-	*Jatka* fishers probably use different gears which could influence catch volume.
Compensation recipient[3]	Binary	Household receives compensation (1) or not (0)	+	Compensation recipients might be expected to say they see an increase in order to show support for the scheme.
Compensation area[3]	Binary	Household lives in area where the compensation scheme operates (1) or in a control area (0)	+	If the compensation scheme is effective then households living in an area where the scheme operated may be less likely to perceive an increase than those outside these areas.
Age	Continuous	Years	+/-	Age, and therefore length of time a recipient has been fishing, may affect the time-frame in which a respondent considers their observations.
Awareness	Binary	Aware of all management interventions (1) or not (0)	+/-	Awareness may influence perceived trends; either through wishing to express support for the intervention or because they live in an area where the intervention has been effective.
Fishing location	Binary	Fishes in sea and river (1) or fishes only in river (0)	+/-	Catch characteristics may differ between those who fish in the sea and those who do not, and so may the impact of management (which is primarily targeting inland areas).
Random effects				
District	Categorical	8 level factor		
Village	Categorical	23 level factor		

[19] These variables were included in separate models due to convergence issues: compensation area was not included where sanctuary and compensation were included, because it is impossible for a household living outside of an area covered by the compensation scheme to also be receiving compensation or living inside a sanctuary.

[20] Juvenile hilsa, officially defined as hilsa of up to 25 cm in length (Islam et al. 2014).

5.3 Results

5.3.1 Theory of change for hilsa management

In Bangladesh, the Department of Fisheries (DoF) has established the HFMAP, which is a suite of management actions for the sustainable management and development of hilsa fisheries (Chapter 3; DoF 2002). Although the word 'conservation' is used in the HFMAP, it should be noted that the recommendations that have been taken forward are focused on sustaining and increasing hilsa production (DoF 2002). They aim to control exploitation patterns (i.e., gear, location, and season) rather than exploitation rates, as is the case in many small-scale fisheries where resources for monitoring and enforcement are limited. These actions, and therefore their potential for ecological additionality, are based on a set of assumptions about hilsa life history, management implementation, and social-ecological context. Here I reconstruct an implied theory of change, based on the regulations that have been implemented from the HFMAP (DoF 2002), showing the theoretical causal pathways between management actions and outcomes, and describing the key assumptions on which they appear to be based (Fig. 5.2). Using the baseline and positive counterfactual from the frame of reference (Chapter 3), KIIs, and fishers' perceptions, I explore the validity of this theory of change by assessing qualitatively the available evidence for each of these underlying assumptions. Following these individual evaluations, I summarise the strength of support for each assumption and the level of importance of each assumption in determining the potential for a given management action to contribute to sustainability (Table 5.5).

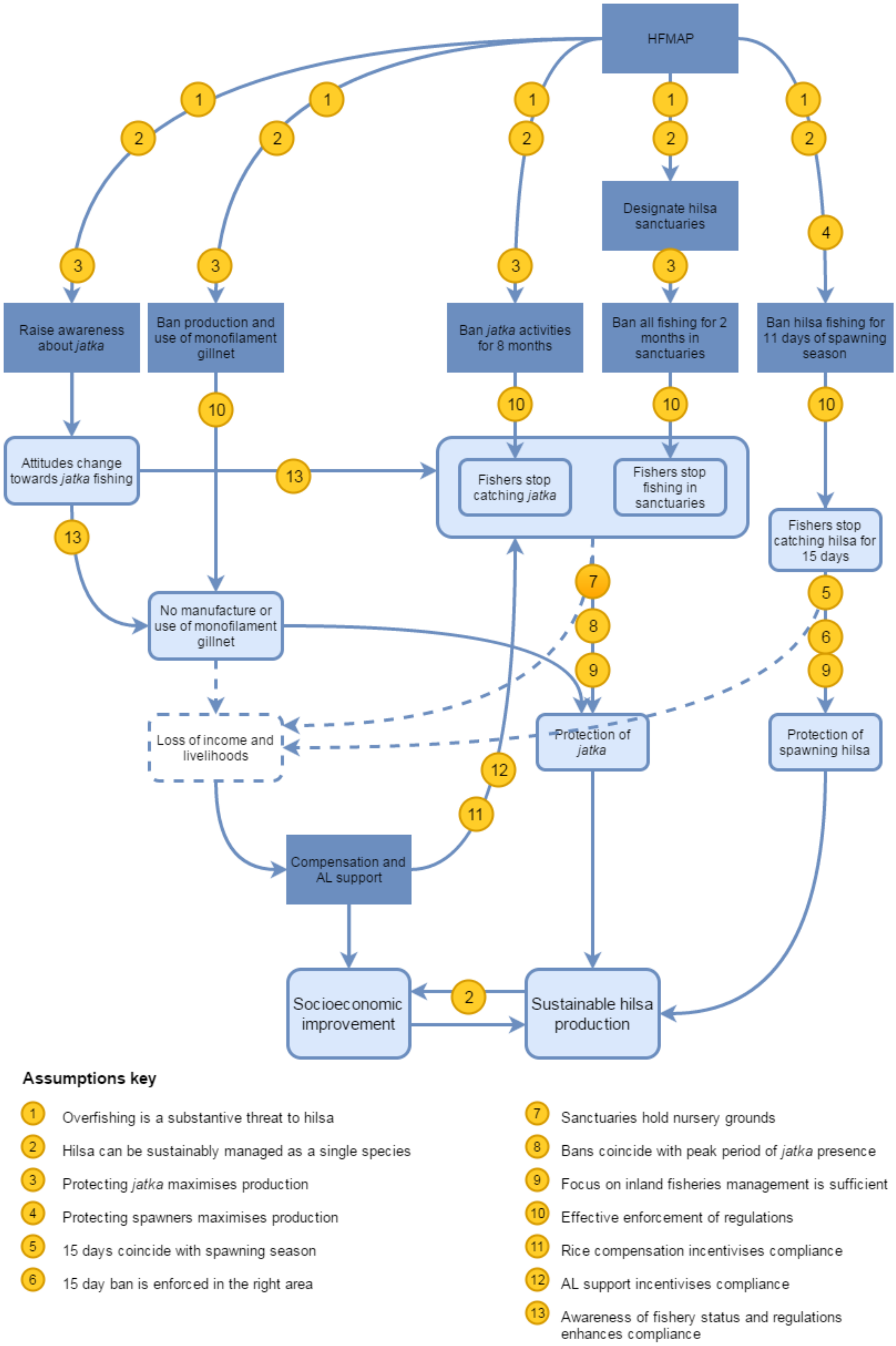


Figure 5.2: Reconstructed theory of change for hilsa fisheries management in Bangladesh. Sharp squares are management regulations that have been implemented, rounded squares are theoretical management outcomes, dotted lines are potentially negative side-effects of management and circled numbers represent the assumptions supporting the logical connections between boxes. AL = alternative livelihood.

Assumption 1: Overfishing is a substantive threat to hilsa stocks

Current hilsa fisheries management is driven by concerns about growth and recruitment overfishing (DoF 2002; Haldar 2004). Chapter 3 presented the limited and often conflicting literature behind these concerns, much of which is based on unreliable national catch and effort statistics. For instance, estimates of catch per unit effort (CPUE) from the artisanal marine hilsa fishery declined between 1984 and 2006 (Mome & Arnason 2007; Sharma 2012). One local NGO KI (LN1) said that 'the Government increases production every year by pen and paper', while an international NGO KI (IN1) said that catch statistics are 'probably quite conservative'. The latter statement is validated by a catch reconstruction of marine fisheries in Bangladesh, which indicates that national catch statistics are grossly underestimated (Ullah et al. 2014). Total hilsa catch is thought by these authors to have been underestimated by 19 per cent since 1950. However, since the majority of hilsa fishing activities occur inland (100,000 vessels estimated inland as opposed to 25,000 in the marine; see Chapter 3) conclusions should not be drawn from marine data only. Although accurate inland effort data is unavailable (inland vessels do not require registration or licensing), the proportion of total hilsa catches coming from the inland fishery has declined dramatically (Chapter 3). It is of course true that, in isolation, a decline in catch rates is not evidence of overfishing (Kolding & van Zwieten 2011). However, the observed and projected growth in human populations in coastal and inland areas will undoubtedly be placing heavy fishing pressure on hilsa populations (Chapter 3; Fernandes et al. 2015).

Length-frequency data analysis has also been used to estimate exploitation levels over time (Amin et al. 2002, 2004, 2008; Haldar 2004; Rahman & Cowx 2008; BOBLME 2010). Although highly uncertain and variable, these studies did show a gradual overall increase in fishing mortality from 1992 to 2009, a decline in size at first capture, and quite consistent evidence of overexploitation (see Chapter 3). Haldar (2004) also found evidence for the overexploitation of *jatka*. Only one of these studies, however, analysed marine and inland populations separately (Rahman & Cowx 2008). This study concluded that both were still under the limit of maximum acceptable effort, but that the inland fishery was more vulnerable to overexploitation than the marine. A productivity-susceptibility analysis (PSA) of hilsa stocks in the Bay of Bengal (BoB) identified a declining trend in most of its productivity parameters (fecundity, catch rates, growth rates, age composition, mortality index, probability of breeding), concluding that despite evidence of recruitment overfishing in Bangladesh, stocks are not depleted or in need of rebuilding (BOBLME 2010, 2011a). Systems dynamics simulation modelling based on population parameters from length-frequency data analysis also suggested that, although stocks are under severe stress and vulnerable to overfishing, they are not in decline under current harvesting practice (Bala et al. 2014). Nevertheless, their projections showed that an increase in fishing pressure on *jatka* and adults could lead to a collapse by 2020. A recent study of marine hilsa production also found estimates of fishing mortality rate in the literature to be much higher than that modelled for MSY, and projected a collapse of the marine hilsa fishery by 2030 without more sustainable management (Fernandes et al. 2015)

Concerns about overfishing are supported by periodic inland surveys of experimental CPUE, with some authors noting a slight decline between 1998 and 2011, although these surveys are not directly comparable and should not be interpreted as time-series (BOBLME 2011b; Rahman & Bhaumik 2012b). Similarly, declines in *jatka* abundance have been reported following government surveys of experimental CPUE (Haldar 2004), although reliability is again questionable; one fisheries scientist (KI R1) even admitted to a culture of downplaying *jatka* abundance to fishers in an attempt to reduce fishing pressure.

Even without reliable stock assessments, it is clear that hilsa are subject to intense and increasing fishing pressure, and KI fisheries scientists from a range of sectors were of the view that this pressure is a great threat to the fishery (R1; R2; IN1; IN3; A2; A3; A4; A5). On the other hand, population modelling in Chapter 4 indicates that hilsa are extremely resilient to this pressure. However, there are numerous environmental factors which may also be driving change in the fishery (Chapter 3; Fernandes et al. 2015). Because of their life history (Chapter

4), hilsa are susceptible to the perturbations in not just marine, but also estuarine and freshwater environments, caused by climate change, pollution, deforestation, siltation and water diversion activities. Although the effects of these perturbations are not well understood in hilsa, species with similar life cycles tend to have strong environmental signals. American shad (*Alosa sapidissima*), for example, have a strong behavioural response to environmental fluctuations; the timing of their spawning migrations varies with latitude and inter-annual variation is correlated with river temperature (Mansueti & Kolb 1953; Quinn & Adams 1996). Because American shad spawn within their migratory pathways and have a brief larval period, the environments experienced by returning adults and their spawn are very similar, and so adults behaviourally adjust the timing of their migration and spawning in order to optimise conditions for their young, in response to environmental variation. Larval survival is thus strongly affected by environmental conditions (Crecco & Savoy 1987). Given the similarities between the early life history of American shad and hilsa, inter-annual fluctuations in abundance of hilsa could be just as much due to environmental change as stock size.

Indeed, a significant inter-annual variability has been observed in projections of hilsa production under projected climate change and associated river flow, nutrient loadings and ocean conditions (Fernandes et al. 2015). Although no causal links have yet been established between drivers of environmental change and changes in hilsa abundance or distribution, production is projected to decline with climate change, even under sustainable levels of fishing mortality (Fernandes et al. 2015). Assuming that fishing is not currently sustainable, the authors concluded that a reduction in fishing mortality to sustainable levels should nonetheless mitigate the impacts of climate change. Yet, without fully understanding the potential impacts of other threats such as industrial pollution and water diversion on hilsa, the extent of habitat loss and degradation that populations could withstand is uncertain.

American shad have experienced declines throughout the United States since the 20th century, which cannot be attributed to one problem, but to a combination of overfishing, habitat degradation, and flow alterations (Klauda et al. 1991). Following several moratoriums, restoration efforts based on fish passage construction and hatcheries have sustained remnant populations but failed to achieve large-scale restoration (Bilkovic et al. 2002; Hasselman & Limburg 2012; Brown et al. 2013). Brown et al. (2013) argued that this restoration depends on the provision of access to historical spawning habitat, which can only really be achieved through removing migration barriers such as dams. Similarly, protection and restoration of hilsa spawning habitat may be a necessity for the long-term sustainability of the hilsa fishery in Bangladesh.

A common theme of all the KIIs was that hilsa needs to be managed holistically in order to address pollution, siltation and climate change. One of the recommendations of the HFMAP was protection and restoration of hilsa habitat through activities such as tree planting, dredging in major river channels, and reducing water pollution (DoF 2002), but according to KIs (D1; R2; A5) these habitat-focused activities are not taking place at the scale required. Efforts to enhance stocks through regulations on fishing may be ineffective if suitable habitat is not available, and so addressing non-fishing-related stresses is crucial. For example, Brown et al. (2013) highlighted the lessons to be learned from dammed coastal rivers in the United States for developing countries like Bangladesh. Even though actions like dam removal might have short-term social, economic and political costs, they can be essential for significant restoration of hilsa habitat in the long term.

Assumption 2: Hilsa can be sustainably managed as a single species

The extent to which single-species management can achieve sustainability depends partly on how sustainability is defined. Historically the sustainability of fisheries was based solely on the species of interest. Now the concept is understood to depend not only on long-term yields of that species, but also on factors such as a healthy ecosystem, the socioeconomics of fishing communities and the needs of future generations (Quinn & Collie 2005; Hilborn et al. 2015). This perspective has paved the way for the development of ecosystem-based fisheries management (EBFM), and there is a now a large body of literature supporting the need to move beyond single-species fisheries management towards an ecosystem approach (Pikitch et al. 2004; Hilborn 2011; Berkes 2012; Jennings et al. 2014). Globally, fisheries are still predominantly managed as single stocks (Skern-Mauritzen et al. 2016), although progress has been made in developing reference points for single stocks in a multi-species context (Law et al. 2014).

In Bangladesh, some KIs (D1; R1; IN3; LN6) voiced assumptions or hopes that the protection of hilsa will have wider impacts on aquatic biodiversity but, as discussed above, limited actions have been implemented for the protection of the ecosystem as a whole. Furthermore, management of hilsa as a single species may not be the most sustainable approach for Bangladesh in terms of maximising total fishery yields for long-term food security. Under current management, a combination of increased fishing pressure and environmental change is projected to lead to the collapse of marine hilsa and replacement by lower value species like chacunda gizzard shad (*Anodontostoma chacunda*), toli shad (*Tenualosa toli*), and various species of tilapia, which are better suited to the projected climate (Fernandes et al. 2015). Multi-species management would therefore be more sustainable in terms of overall biomass production (Kolding & van Zwieten 2011; Kolding et al. 2015), despite potentially incurring some economic losses. KIs (LN4; M2) pointed out that morphological similarities between hilsa, chacunda, and toli shads mean that the latter two species are often confused for and sold as hilsa in markets, and so these species should be managed together, regardless.

Assumption 3: Protecting jatka maximises production

The selective fishing paradigm has long been a cornerstone of fisheries management, guiding spatial and temporal restrictions on fishing, and gear and mesh size regulations, which intensify the already selective nature of fishing (Garcia et al. 2012). In Bangladesh the DoF aims to increase and sustain the production of hilsa, largely through actions focused on the protection of *jatka* (Fig. 5.2). It has banned the monofilament gillnets used to catch *jatka*, has introduced spatial and temporal fishing bans for their protection, and provides rice compensation, alternative livelihood support, and awareness building for *jatka* fishers. This convention of limiting exploitation of small and immature fish is underpinned by both yield-per-recruit theory and the spawn-at-least-once principle (Froese 2004) but, as explained in Chapter 4, when multi-species interactions are considered this convention becomes questionable, particularly for developing countries where small fish frequently play a prominent role in food security (Pope 1991; Kolding & van Zwieten 2011; Law et al. 2014). Furthermore, its implicit assumption that all protected immature fish will contribute to future catches is rarely true; recruitment is heavily influenced by extrinsic factors such as environmental variability and the age range of spawning populations (Cardinale & Arrhenius 2000; Wieland et al. 2000; Szuwalski et al. 2015). In the Connecticut River, for example, American shad year class strength is established before the juvenile phase, and recruitment is thought to be regulated more by river flow and temperature than stock size (Crecco 1985; Crecco & Savoy 1987). The impact of environmental conditions on egg and larval survival might also be playing a more critical role in determining hilsa recruitment than *jatka* survival. Observed and projected warming temperatures, reduced freshwater flow, and water pollution therefore have the potential to undermine any regulations designed to increase production through the protection of *jatka* (Chapter 3).

Systems dynamics simulation modelling has provided some validation for the approach of focusing management on the protection on *jatka* in Bangladesh. Results indicate that the overfishing of *jatka* leads to a gradual decline in stock, and that the juvenile life stage is the most critical in terms of fishing mortality – although the study did not account for external environmental factors (Bala et al. 2014). In contrast, running a simple age-structured population model for hilsa under a range of fishing mortalities indicated that targeting *jatka* (represented by fish aged < 1 year) is actually consistently less damaging to population biomass than targeting mature adults or fishing across all age classes (see Chapter 4). Although differences in impact between the selectivity regimes were small, the results indicate that hilsa are more sensitive to exploitation rate than to size selectivity.

A further problem with selective fishing is that it alters the composition of a population or community (Hsieh et al. 2010; Zhou et al. 2010; Garcia et al. 2012). As discussed in Chapter 4, the selective removal of large and old individuals over small means that earlier maturation and smaller adult body size become evolutionarily advantageous, with potentially long-term consequences for age (size) structures and sex ratios (Law 2000; Kuparinen et al. 2009; Law et al. 2013). Given the superior breeding potential of large and old individuals, it follows that regulations focused on the protection of juveniles may therefore have negative impacts on abundance (Anderson et al. 2008; Kolding & van Zwieten 2011; Rochet & Benoit 2012) – although some authors maintain that truncations in age (size) structure are caused by excessive fishing rather than minimum-size limits (Froese et al. 2016). While a decline in size at first capture has been observed in the hilsa fishery (Haldar 2004), this has not been causally linked to any specific fishing or management activity.

Studies of balanced harvesting propose that moderate fishing across the widest possible range of species, stocks, and sizes in an ecosystem, in proportion to their natural occurrence, should maintain relative size and species composition, and increase total yields and biomass (see Chapter 4). Empirical evidence supporting the concept is limited and it has been criticised as an unrealistic strategy for management (Jacobsen et al. 2014; Breen et al. 2016; Froese et al. 2016). Yet, size-spectrum models have been used to usefully explore the ecosystem consequences of different management strategies, and indicate that management focused on the protection of juveniles may not maximise overall production, or indeed minimise the negative impacts of fishing (Law et al. 2014; Andersen et al. 2016).

Assumption 4: Protecting spawners maximises production

Management actions do not focus on the protection of *jatka* alone. The DoF has also banned hilsa fishing for 15 days during peak spawning season, with the aim of minimising disturbance to spawning (Chapter 3). Again, this action is supported by the spawn-at-least-once principle, in which letting all fish reproduce is expected to maximise subsequent recruitment (Froese 2004). Modelling indicates that if fish are not harvested until the completion of their first spawning season, and if every spawner produces at least one replacement spawner then, assuming successful recruitment, a stock will not collapse at any fishing mortality (Myers & Mertz 1998). The same criticisms of this theory apply here; recruitment is not always successful and may be heavily influenced by environmental conditions, which play an important role in recruitment for species with similar life cycles (Crecco 1985; Crecco & Savoy 1987). The impact of these environmental conditions on egg and larval survival might play a more critical role in determining hilsa recruitment than the size of spawning stock and so observed and projected warming temperatures, reduced freshwater flow, and water pollution have the potential to undermine this impact of this regulation (Chapter 3). For many species the age structure of the current spawning population also complicates the stock-recruitment relationship, since younger spawners usually have the lowest fecundity and larval survival (Cardinale & Arrhenius 2000; Marteinsdottir & Begg 2002) – although this has not been demonstrated specifically in hilsa.

Nevertheless, there are evidence-based arguments supporting the need to recognise fish spawning aggregations as a focal point for management and conservation (Erisman et al. 2015). The timings and perhaps even locations of spawning hilsa aggregations are predictable for local fishers (Ahsan et al. 2014), making them attractive times and sites for fishing with an increased risk of overexploitation, but also easier to enforce. In theory, banning hilsa fishing completely during the peak spawning season does reduce this risk, and in comparison to year-round restrictions on fishing, avoiding an area for a shorter period of time can be very efficient. On the other hand, the reduction in availability of or access to suitable spawning grounds for hilsa in Bangladesh may reduce the potential for this management action to have impact (Chapter 3).

Assumption 5: The hilsa fishing ban (for spawners) coincides with peak spawning season

The hilsa fishing ban for the protection of spawners was, until 2015, implemented annually for the 5 days before and 5 days after the full moon of *Bara purnima,* in the Bengali month of *Ashvin* (which falls in October; Chapter 3). The assumption that this ban coincides with peak spawning season firstly depends on it falling within peak spawning season, and secondly on it being long enough to allow sufficient spawning. Investigations of spawning seasonality using the gonadosomatic index method[21] indicate that spawning occurs year round with a peak in September and October (Hasan et al. 2016; Rahman et al. 2012a). There is some evidence to suggest that there may also be a distinct and smaller winter (January) spawning stock, which has a smaller migratory size and lower fecundity (Quddus 1982; Rahman et al. 2012a; Ahsan et al. 2014), and one KI (D5) suggested that the ban might need to be replicated in January, but Hasan et al. (2016) challenged the existence of any secondary peaks. There is evidence from other fish species that female age or size can affect the timing and duration of spawning (Hsieh et al. 2010).

The placement of the ban around the full moon had previously been justified with the traditional understanding that spawning peaks in each lunar month during the spring tide (Rahman et al. 2012a). Again, however, Hasan et al. (2016) found no evidence for this. One local NGO KI (LN2) also noted that the ban period was systematically designed to be aligned with traditional beliefs of the fisher communities, and linked the original 11-day ban with the Hindu festival *Durga puja*, which falls during the same period. The placement of the ban could have been motivated more by potential to reduce political conflict than by scientific evidence. A range of KIs (D1; A4; A5) reported that the peak is shifting towards winter – a trend which is not surprising, given warming temperatures in Bangladesh (Chapter 3). However, 100 per cent of interviewed fishers said that the ban period (11 days at time of interview) fits the spawning

[21] (gonad weight/body weight)*100

season. There also claims that the timing and duration of the ban often varies locally, indicating a lack of communication between scientists and authorities (Islam et al. 2016).

On the basis of research by Hasan et al. (2016), in 2015 the ban was extended to 15 days and shifted to the 3 days before and 11 days after the full moon. This should further minimise disturbance to spawning during peak season – although since recommended extensions from KIs ranged from 15 days (R1) to 30 days (A4; A5), it may still be too short. One KI said that 'there needs to be some grace period'; i.e., the ban period should extend either side of the peak spawning period (IN4). This will be particularly important if there is substantial inter-annual variability in spawning period, as has been observed in American shad (Crecco & Savoy 1987). Furthermore, no scientific basis has been provided for the DoF's decision to shift the timing of the ban in relation to the full moon, so it is unclear whether it is scientifically motivated.

Assumption 6: Enforcement of the hilsa fishing ban (for spawners) is targeted to the right area

Research indicates that hilsa spend most of their adult lives in the Bay of Bengal, with the majority migrating upstream to spawn in freshwater (Chapter 4). Although the 15-day hilsa fishing ban is officially implemented throughout the country to avoid confusion (KI D2), due to resource limitations monitoring for compliance is targeted within a 7000 km^2 area (see Fig. 5.3) that is thought to cover four important spawning grounds, identified through experimental fishing (the Moulvir char, Monpura, and Dhalchar areas of Meghna River and Shahbazpur Channel; Haldar 2004). But hilsa are not strictly anadromous; eggs and larvae have been found in coastal waters too (Blaber et al. 2003a; Rahman et al. 2010, 2012a) outside the area of enforcement. There is also evidence to suggest that fish of different age classes spawn in different areas (Hsieh et al. 2010), and so the spatial distribution of hilsa spawning aggregations may be much more complex than current research indicates; for instance it may be that megaspawners could be protected by focusing on a specific area. It has also been reported that spawning grounds are shifting seawards (DoF 2002), a shift probably driven by a combination of climate change and anthropogenic activities that are likely to be reducing the quality and accessibility of inland spawning grounds (Chapter 3). More research is needed to validate the concentration of enforcement effort in spawning grounds that were identified as important over one decade ago.

Assumption 7: Hilsa sanctuaries hold nursery grounds

When larvae become *jatka*, they are traditionally understood to find nursery grounds in the rivers and coastal areas, where they stay for 5-6 months, before migrating back to the BoB (Chapter 4). Four hilsa sanctuaries were therefore designated in the Meghna River and inshore marine area, in 2005, in which all fishing is banned seasonally for the protection of *jatka* (Fig. 5.3). They were designated on the basis of a study that identified them as major nursery grounds from the abundance of *jatka* found in experimental fishing surveys (Haldar 2004). A fifth sanctuary was designated in the lower Padma River in 2011, on the basis of continued study (Rahman & Bhaumik 2012b), and a BFRI scientist revealed that discussions around adding a sixth sanctuary in Barisal are ongoing (R1). Since the marine phase of hilsa is much less well studied than the freshwater, it is unclear whether there are significant unprotected nursery grounds further offshore.

Key themes that emerged from KIs were deterioration of water quality in the sanctuaries and shifting distribution of *jatka* (D2; R1; R2; LN2; IN2; IN4; IN6; A3). One KI scientist (R1) said that sanctuary designation is keeping pace with this shift, and indeed the addition of a fifth and potentially sixth sanctuary indicates that some monitoring and adaptive management of the sanctuaries is taking place. However, there has been no degazetting, and there are concerns in the literature that the sanctuaries are not keeping pace with environmental change (Islam et al. 2016). For example, water diversion activities, constantly changing river morphologies and recent upward trends in sedimentation may be restricting access to sanctuary areas and reducing water quality (Chapter 3). Islam et al. (2014) reported that a sandbar near Chandpur (Fig. 5.1) has restricted hilsa migration, resulting in low hilsa numbers in surrounding areas that are still within sanctuary boundaries. Pollution is another threat to the sanctuary areas, through its influence on water quality (Chapter 4). In the Andharmanik River (Fig. 5.3), site of the Patuakhali sanctuary (Fig. 5.1), one KI (LN2) said water quality is so low that it is now known as the 'dead river'. A recent assessment of physical, chemical and biological parameters of hilsa habitat also found the river to have significantly higher levels of nitrogenous compounds, higher levels of sedimentation and lower levels of chlorophyll a than other sites sampled within the study area of this thesis, as well as a different plankton community composition (Hasan et al. 2015). Although the study concluded that these parameters were still within acceptable limits, it provides some validation for anecdotal claims that the sanctuary is no longer suitable hilsa habitat. Potential drivers of change in these sanctuaries – pollution, water diversion activities, siltation, climate change and deforestation – are all trending upwards (Chapter 3), and have the potential to undermine any protection that the sanctuaries may theoretically offer. One DoF KI (D1) said that the Andharmanik river sanctuary should certainly be moved, but implementation of such a measure would be unpopular with local fishers and therefore politically challenging. An NGO KI (LN6) said that a more realistic management option would be to introduce measures to improve the quality of the water, though it might be more expensive and controversial with other industries.

Assumption 8: Fishing bans for the protection of jatka coincide with peak jatka abundance

All fishing is banned in the hilsa sanctuaries from March to April (originally February-March), apart from in the Andharmanik River, where it is banned from November to January (Fig. 5.3). There is also a nationwide *jatka*-fishing ban, first enacted between November and April each year, later extended to May and then to June, when it was also extended to all activities related to *jatka* (catching, transportation, marketing, selling and possession), and then to July (9 months long). The desired outcome of these bans is increased recruitment into the fishery (DoF 2002), but this depends on whether the timings fit the life cycle of hilsa.

Although *jatka* are present throughout the year, experimental fishing and analysis of commercial catch found them to be most abundant from January to May in rivers, with a peak from March to April, and from December to January in coastal areas, when larger-than-usual *jatka* are caught (Haldar 2004). This is the only evidence available to justify the timing of the bans on all fishing in sanctuary areas. One KI (R1) pointed out that there are very few *jatka* seen in November (reportedly due to pollution) and so the Andharmanik River ban needs shifting forwards, but this process will take two to three years. A range of KIs (R1; LN2; IN4) referred to a temporal shift in *jatka* availability towards later in the year, which is consistent with the shift of the March to April ban from February to March. Given the suspected climate-driven shifts in spawning seasonality, it is possible that environmental change is affecting the timing of *jatka* availability (Chapter 3), in which case the bans on fishing in sanctuaries may not be protecting a substantial proportion of *jatka* and therefore require reinvestigation.

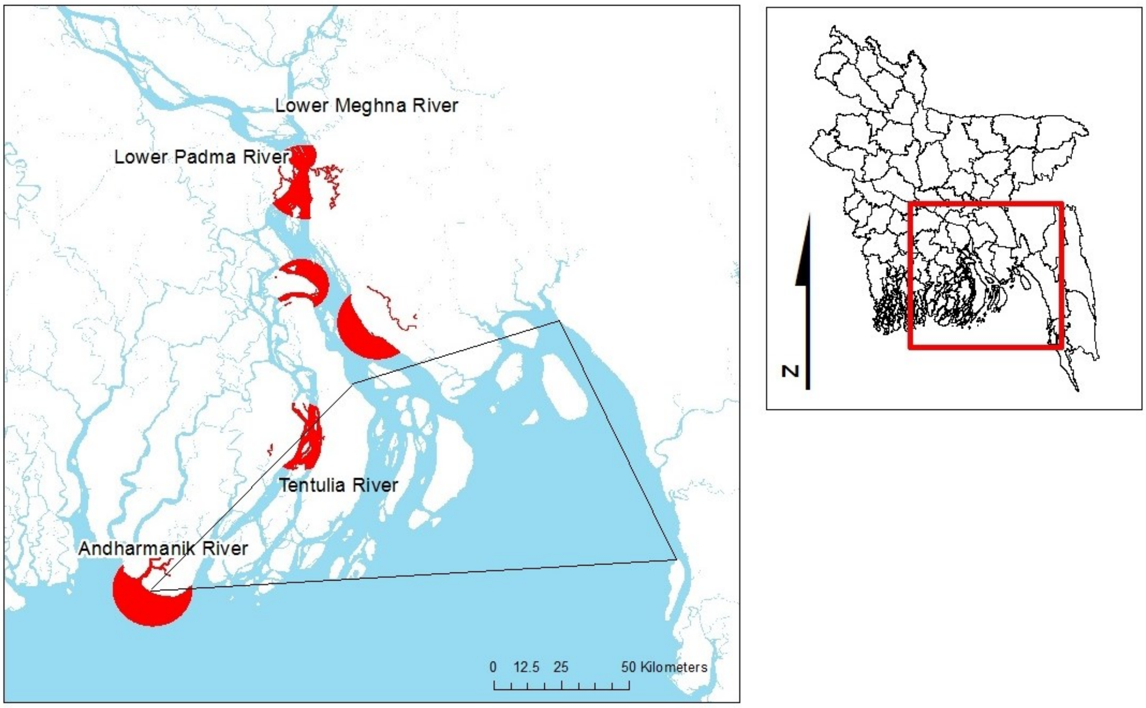


Figure 5.3: Map showing sanctuary areas (red) and rivers flowing into the Bay of Bengal (blue). All fishing is banned in the sanctuaries from March to April, apart from the southernmost sanctuary, a 40 km stretch of the Andharmanik River where fishing is banned from November to January. Black polygon demarcates important spawning area where enforcement of 15-day hilsa fishing ban is targeted during peak spawning season.

The widely accepted continuous presence of *jatka* in rivers and coastal areas is backed up by the almost continuous pattern of recruitment observed through length-frequency data analysis

(Haldar 2004; Rahman & Cowx 2008) – although these studies are conflicting. Haldar (2004) found a major peak in recruitment in June and July (although the peak for males was May to July and for females March to May and July to September), and this pattern continues to be reported by Government scientists (e.g. Rahman et al. 2012a). Rahman & Cowx (2008) analysed marine and inland populations separately and found major and minor peaks: in rivers, the major peak was from March to May and the minor from November to January, and in the marine environment they were shorter (March to April and January to February). The length of the ban on all *jatka* activities reflects the fact that they are present throughout the year, and the extensions that have been made indicate that management is adapting to changes in abundance and new knowledge. Some KIs from academia and NGOs (LN5; A4; A5) said that there should be a complete ban on *jatka* activities throughout the year, but this may not be a realistic proposition given the high dependence of the poor on *jatka* (Chapter 6).

Assumption 9: Focus of management on inland fisheries is sustainable

Hilsa management is strongly focused on the inland fisheries (riverine and estuarine areas). This focus probably grew from traditional understanding about *jatka*, which – through ease of implementation and the historical focus of fishing in inland areas – has mostly been developed through inland research. Although the inland fishery is estimated to have four times the capacity of the marine fishery in terms of numbers of fishing vessels, the marine fishery now produces the majority of hilsa (Chapter 3). Nevertheless, the only piece of legislation directly affecting hilsa fishers and implemented in the marine environment is an amendment of the Marine Fisheries Ordinance and Rules, which bans all fishing by all vessels in the marine fisheries between May 20th and July 23rd each year (MoFL 1983). Even those regulations that apply nationwide (*jatka* rules and the 15-day hilsa fishing ban) are not enforced offshore (Chapter 3). Given the fact that *jatka*, eggs and larvae have been found in the marine environment, and given the limited understanding of hilsa life history (Chapter 4), focusing management on inland fisheries may not be providing sufficient or efficient protection.

Assumption 10: Regulations are effectively enforced

Effective enforcement is defined here as enforcement that promotes near-complete compliance, which should be conducted in a socially just way. There is very little evidence for effective enforcement of regulations; in fact, the limited literature highlights a lack of enforcement and low levels of compliance (Islam et al. 2016), issues that also arose as key management constraints in KIIs (L1; L2; D4; D2; LN1; LN3; LN4; IN1; IN2; IN3; F1; F2; A2; A4). One local academic said that the management approach is good, but there is no enforcement and the situation is devastating for inland fisheries (KI A1). Even a local Government KI (L2) said 'There are regulations, but in a practical sense these are not happening'.

Despite the ban on monofilament gillnets, their use is still widespread (Islam et al. 2016). According to local KIs (D5; LN1; LN5; F1; F2; M1; M3), *jatka* fishing continues throughout the *jatka* ban, particularly at dawn and dusk when the authorities are not on the water, and although markets do close, *jatka* continues to be sold on the riverside instead, with coastal surveillance pushing landing sites further inland. These reports do nevertheless indicate that enforcement is having some impact on behaviour and thereby imposing some small costs on fishers.

When hilsa fishers were asked about compliance with sanctuary fishing bans, the majority said that few fishers comply (66 per cent said that few or no compensation recipients comply, while 78 per cent said that few or no non-recipients comply) and none said that all fishers comply (Fig. 5.4). Furthermore, 40 per cent of respondents said that they fish illegally as their main coping strategy during closed seasons. One industry KI in Chandpur district (M3) said that there is about a 70 per cent level of compliance, while Islam et al. (2014) reported a 92-95 per cent level of compliance in the same area.

Local KIs (LN1; LN2; M1; M3; M4; M5; F2) said that, in contrast, the nationwide hilsa fishing ban for spawners is generally complied with, despite the lack of compensation during this period. This could be due to the short time-period or better enforcement, or it could be related more to its roots in the traditional beliefs and activities of Hindu fishing communities at that time than to enforcement of legislation (although it should be noted that hilsa fishers are now majority non-Hindu). Another possibility is that the cyclone season (October to November) deters fishers during this time anyway (DoF 2002).

Numerous factors limit the effective enforcement of regulations. Enforcement of *jatka* regulations is targeted to 152 *upazilas*[22] where *jatka* are abundant, particularly areas thought to be important nursery grounds (Chapter 3). However, there is a lack of human and financial resources for proper surveillance (DoF 2002; Islam et al. 2016), which is even admitted within the DoF itself (D3; D4; D5). The DoF is supported by police, local administrations, and – on the coast – the Air Force, Border Guard, Navy, coast guard, and Rapid Action Battalion, but NGO KIs (LN4; IN3) said that there are not enough patrol boats and there is a lack of surveillance inland. Mobile courts are operated to deal with offences on site (between 800 and 1000 were reported per year between 2011 and 2014) but effective operation is still limited by resources (Islam et al. 2016). Islam et al. (2016) also noted that sanctions (numbers of prison sentences, fines, court cases etc.) have increased from 8,262 (for *jatka* offenses) and 16,525 (for hilsa offenses) in 2011/2012, to 22,077 and 33,050 in 2013/2014, but it is not clear whether this reflects a decrease in compliance or an increase in enforcement efforts.

Secondly, the sanctions are described as an ineffective deterrent by a range of KIs, including local Government officials (L1; L2; LN1; LN4; IN1; F1; F3). Local NGO KIs (LN1; LN4; LN5) said that inflicting heavy fines or long jail sentences on fishers simply perpetuates the cycle of debt that leads them to break the rules in the first place, and when nets and catches are burned, fishers are forced to take new loans to repay their debts or buy new nets. A central Government official (KI C1) said that they try to give jail time instead of fines because most fishers cannot pay the fines anyway, but an NGO KI (LN4) said that the Government has a blind administration which confuses enforcement with harassment. This leads on to the issue of corruption, which is

[22] Subdistrict

deeply rooted in the institutional framework of Bangladesh (Chapter 3); authorities were widely reported by KIs, including local Government and DoF KIs, to take bribes from fishers in exchange for ignoring illegal activities (L1; L2; D3; F2; F3). Furthermore, some industry KIs (M1; F2; F3) pointed out that fishers are usually financed by middlemen who are unaffected by the sanctions and continue to send labourers out to catch *jatka* during the bans. They also said that the sanctions do not deter seasonal or occasional fishers (i.e., those who are not fully dependent on fishing) because the cost of a net can easily be earned back with one night of *jatka* fishing.

There is also an external barrier to effective enforcement, which is the lack of access to appropriate financial products for fishers. Although microcredit is becoming more accessible, fishers still borrow from informal sources and are required by all sources to pay interest during the bans on all fishing in the hilsa sanctuaries, leaving them with little choice but to go fishing anyway (Uraguchi & Mohammed 2016). On the other hand, even if bribes are being paid and new nets and debts and being taken on, this is evidence of at least some cost being imposed on fishers by enforcement.

Assumption 11: Rice compensation incentivises compliance with the sanctuary fishing bans

In recognition of the socioeconomic hardships imposed by the fishing bans, in 2004 the DoF introduced the *jatka* fisher rehabilitation programme (see Chapters 3 and 6 for details). This is largely based on the provision of rice compensation to the poorest fishers affected by the bans in hilsa sanctuaries. By reducing the vulnerability of these fishers, in theory the compensation should incentivise compliance with fishing bans (Chapter 2). There are, however, a number of flaws in the scheme which may limit effects on compliance. Firstly, there are issues with the distribution of compensation that probably limit the perceived fairness and legitimacy of the scheme – factors which usually play a role in determining compliance with rules (Chapter 6). Much of this can be attributed to limited resources and to political interference (Siddique 2009; Haldar & Ali 2014), which can in turn be linked to a rise in constituency-based politics and associated competitive pressure (Hossain 2007). Moreover, despite the fact that all fishing is banned in the sanctuary areas, only *jatka*[23] fishers are compensated, which could create a perverse incentive for other people to claim to be *jatka* fishers or to start targeting hilsa, some of whom may otherwise have fished other species or not fished at all (Corrêa et al. 2014; Islam et al. 2016). An identification card scheme is currently being phased in to reduce the potential for non-*jatka* fishers to claim compensation (D1), but it is unclear how these are allocated and their distribution could entail the same problems. A local NGO KI (LN4) voiced concerns about political bias in their allocation. Similarly, even though some sanctuary areas appear to be becoming environmentally unsuitable for *jatka*, fishers living in these areas continue to receive compensation for not fishing (Islam et al. 2016). This problem is again deeply rooted in constituency-based politics; if a Government official were to retract compensation from fishers in a particular area, the decision would be met with resistance from communities in that area and possibly lose him political support (Hossain 2007).

Islam et al. (2014) noted that the exclusion of boat owners and other middlemen from the scheme might limit the impact of the scheme on compliance because they are the individuals with the influence and control over fishers. The boat owner KI (F1) was of a different opinion, however, claiming that they never employ fishers during fishing bans because they understand the benefits. KI F1 said that because they are the 'brothers of fishers', fishers will listen to them more than they do to the Government, but they do not have the capacity to exert the influence they desire. Whether or not these are genuine statements, and whether they reflect other boat owners' opinions, is of course unknown.

There are also issues with the nature of the compensation that may damage attitudes towards the scheme and affect its impact on compliance. The rice was widely regarded by non-Government KIs as inadequate and in some ways inappropriate (LN1; LN4; LN5; IN1; A4; A5). Not only are household sizes not taken into account, the lack of resources for distribution and political interference means that households do not receive the full 40kg which they are

[23] This term is often used in Bangladesh to refer to very poor hilsa fishers, rather than strictly fishers who target *jatka*.

allocated (Haldar & Ali 2014; Islam et al. 2016). A key limiting factor identified by local and international NGOs (LN2; LN4; LN5; IN4) was that most poor fishers are stuck in cycles of debt which pass from generation to generation, so even if they receive rice compensation they are still forced to fish to repay their debts, reducing the opportunity for compensation to enable compliance. Furthermore, rice does not address the lack of protein which households may experience during the ban, which may force them to fish anyway (Islam et al. 2016). A local academic (KI A4) said that a fisher can catch two to three months' worth of earnings in one night if they target *jatka*, and that 'people are not greedy; they have to do it.' Timing may also be a limitation; whereas the *jatka* ban lasts from November to July, compensation is only provided from February to May, and there is little evidence to support the expectation that this covers the peak period of *jatka* abundance (Haldar 2004). *Jatka* are available all year round, and despite receiving compensation in February, a fisher may still feel compelled to catch *jatka* in January.

Finally, the incentives are undermined by the poor enforcement of regulations described above (Islam et al. 2016), which means that there is a lack of conditionality on behaviour (see Chapter 2). One international NGO KI (IN4) said that the rice is excess from the pre-existing Vulnerable Group Feeding (VGF) programme, describing the scheme as 'nothing more than a social safety net'. Nevertheless, fishers reported that they perceived levels of compliance with sanctuary closures to be significantly higher among compensation recipients than non-recipients (χ^2 = 26.5, df = 3, $p < 0.001$; Fig. 5.4). It is therefore possible that despite the lack of conditionality, the scheme could be having a small impact on compliance. One academic KI (A5) said that the scheme is contributing to a gradual decline in the numbers of illegal fishers in recipients and non-recipients, both groups slowly becoming more aware and looking for alternatives. A local NGO KI (IN5) said 'even if it is only 50 per cent effective it is good.'

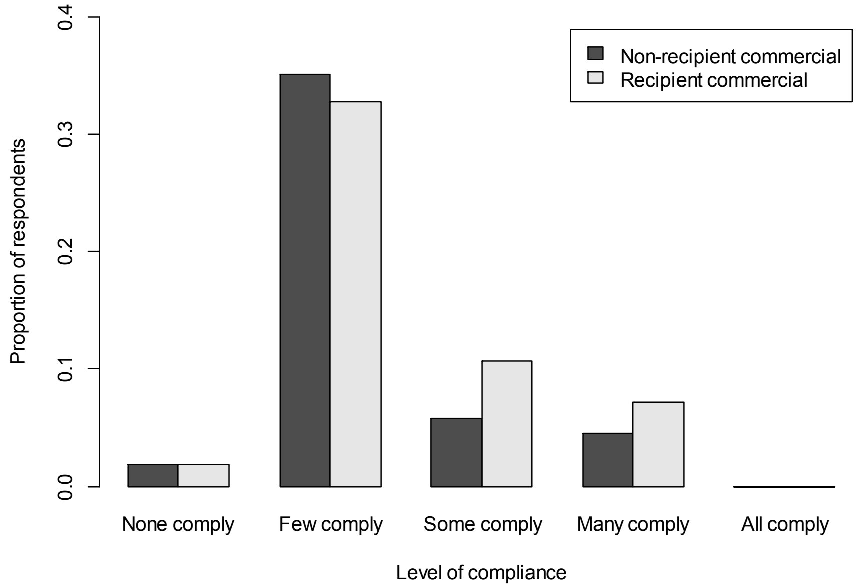


Figure 5.4: Graph showing level of compliance with sanctuary closures by compensation recipients (n=673) and non-recipient fishers (n=747), as perceived by all respondents. Sample sizes were not the same because not all respondents answered each question.

Assumption 12: Alternative livelihood support incentivises compliance with regulations

In 2008 the *jatka* fisher rehabilitation programme was extended to provide a smaller proportion of households with alternative livelihood training and support (e.g. rickshaws, vans, livestock, and grants) to enable them to diversify their income sources. The aim was to improve the livelihoods of fishers affected by *jatka* regulations, with the rationale that this should make fishers more able and willing to comply with regulations. Unlike the rice compensation, it should provide an alternative means of making money, not just during the ban period, but for the entire year.

Very little information is available for the validation of this assumption, which is probably related to the fact that the number of households receiving alternative livelihood support is very low; only 0.5 per cent of the numbers of households currently receiving compensation receive alternative livelihood support, and numbers receiving alternative livelihood support have been declining (see Chapter 3). This decline is probably due to the fact that it requires the allocation of extra financial resources – unlike the rice compensation scheme, which runs through the VGF. Nevertheless, there have been calls for an increase in coverage, and it has been described as 'highest priority' in the literature (Siddique 2009; Rahman et al. 2014b; Islam et al. 2016). A DoF KI (D2) said that they would like to increase coverage, but that it will 'take time and more money than we have'. In a recent choice experiment, compensation recipients showed a higher preference for alternative livelihood support than rice compensation, while non-recipients showed no preference for one over the other – a finding that has been linked to the dissatisfaction with the amounts of rice that are currently received (Dewhurst-Richman, Mohammed, Ali, *et al.*, 2016). However, a number of local NGO KIs (LN3; LN4; LN5) said the support is not useful to fishers since they lack the skills required to use it – a failing that has been attributed to a lack of needs assessment and stakeholder engagement with fishers (Haldar & Ali 2014; Islam et al. 2016). An understanding of people's needs, aspirations, and the factors influencing livelihood choice is a necessity for the use of any livelihood-focused intervention as a direct behaviour-change tool (Wright et al. 2016). Although an in-depth exploration of this assumption is beyond the scope of this chapter, Wright et al. (2016) provide evidence to suggest that the kind of alternative livelihood support given in Bangladesh may not necessarily reduce the need and desire to exploit hilsa, and certainly not if the 'alternative' is an inappropriate substitute.

Assumption 13: Awareness of fishery status and regulations enhances compliance

Through boat rallies, mass media and the distribution of leaflets and posters, the DoF raises awareness about a) the importance and status of the hilsa fishery, with an emphasis on *jatka*; and b) *jatka* regulations (DoF 2002; Chapter 4). One slogan used is '*jatka* are the future, so if *jatka* are caught, hilsa are lost'. Members of a boat owners association and a national fisher association said in KIIs (F1; F3) that they are involved in these awareness-raising activities, helping to circulate the rules and regulations down to the community level. A DoF KI (D2) described awareness as a 'principal requirement for sustainability [of the fishery]' and local NGO KIs (LN4; LN5) stated a need for more awareness raising activities.

Awareness of regulations and an understanding of the rationale behind them have been empirically demonstrated to influence compliance in fisheries and marine protected areas (McClanahan et al. 2005; Read et al. 2011; Velez et al. 2014; Thomas et al. 2015). Non-compliance can be unintentional (if an individual is unaware of the rules) or uninformed (if an individual is unaware of the consequences of breaking the rules), but it may also be intentional and informed (Read et al. 2011). What reduces the risk of intentional non-compliance is acceptance of the regulations, which depends not only on awareness and understanding of their importance, but also on their perceived fairness and legitimacy (Sutinen & Kuperan 1999; McClanahan et al. 2012). 74.5 per cent (567) of respondents in the household survey were aware of all management interventions in the questionnaire, with 100 per cent aware of the *jatka* ban and 15-day hilsa fishing ban, and 74.5 per cent aware of the hilsa sanctuaries. All of the 25.5 per cent of respondents who were not aware of the hilsa sanctuaries lived outside sanctuary areas. It is clear therefore that fishers are aware of the existence of relevant regulations, but the rationale behind and importance of the regulations might be less well communicated. Since literacy rate is poor, education through the medium of leaflets and posters could be failing to reach all fishers (DoF 2002). Even if the rationale is communicated, then its influence on compliance is probably limited by the socioeconomic position of the fishers. Since many are stuck in cycles of debt, an understanding of the long-term benefits of protecting *jatka* may not be enough to stop them fishing during ban periods. As one local NGO KI said, 'they have to break laws because they are vulnerable…awareness is one thing, but they need to have other options' (LN4). It is interesting that awareness of the hilsa ban for spawners is so high when awareness-raising activities are focused on *jatka*, and perhaps this is evidence to support the claim that this regulation was tailored to local fishing traditions.

5.3.2 Perceptions of recent trends and management impact

Following an increase in exploitation and fishing mortality rates through the 1990s, a decline in both was observed between 2003 and 2009, together with an increase in length at first capture, while officially reported catches continued to increase (Chapter 3). Although these trends cannot be directly attributed to any management intervention, they do indicate a possible reduction in fishing pressure from around the time of their introduction. Furthermore, the majority of KIs (C1; L1; D1; D2; D3; D5; R1; R2; LN2; LN4; IN1; IN2; IN3; IN4; IN5; IN6; A1; A2; A4; A5) were of the view that the current management approach is having a positive impact on hilsa production. Reported trends in production did vary however: KI D5 stated that hilsa production has increased in the last 10 years, whereas KI M5 stated that '10 years ago there were more hilsa'. One international fisheries expert (KI A2) said 'the top-end approach appears to be working for hilsa'. A local NGO KI (LN4) said that 'the benefit from the [sanctuary fishing] ban is clear'. In-country scientists have also claimed that protection of adult and juvenile stocks has led to increased CPUE in the hilsa fishery, but there are no published data to support this (Rahman & Bhaumik 2012b) and KIs from academia and NGOs (LN4; IN4; A2) said they expected these claims were probably fabricated. Despite the blocked migratory pathways and degradation of inland hilsa habitat reported in the literature, a DoF KI (D6) said that hilsa migrations are now being seen further upstream than usual, for example in Syhlet and Kushtia (north of Dhaka) – evidence that its distribution may be expanding again. But, again, these observations have not been documented, and could be a result of environmental change rather than management.

Ninety-seven per cent of households said that both hilsa catch and abundance had increased over the last five years. When asked about current trends in hilsa catch, without a time frame, 84 per cent of respondents still said it was increasing from year to year, while 12.1 per cent said it had stayed the same and 3.4 per cent said it was decreasing. Modelling revealed a small positive effect of living inside a compensation area on whether or not fishers perceived an increase in hilsa – although it had a low relative importance (0.38) and weak support for inclusion in top models (Table C.1a; Table C.2). Support for inclusion of sanctuary area in top models was much stronger (Table C.3) and its relative importance was 0.93 (Table 5.3); fishers living in or around sanctuary areas were significantly more likely to report an increase in catch volume than those who live outside sanctuary areas. Compensation had a very low relative importance in these models (0.22), and none of the other fixed effects were very important, apart from age (relative importance 0.51); older fishers were less likely to report an increase in catch (Table 5.3). Plotting the BLUPs for each district and each village within district showed a significant effect of geography on the probability of reporting an increase in catch, once fixed effects were taken into account. Households in Bhola and Laxmipur districts were most likely to report an increase, whereas those in Chandpur and Patuakhali were least likely (Fig. C.1). Variation between villages within a district was even stronger: households in Baushia and Gourabdia (Barisal district) were significantly more likely to report an increase, whereas households in Nizampur and Mewrapara (Patuakhali district), Kutubpur (Barisal district) and Uttar Gobindia (Chandpur district) were significantly less likely (Fig. 5.5a).

Table 5.3: Results for GLMMs of probability of reporting an increase in hilsa catch volume from year to year, showing the full model-averaged coefficient estimates (standard error) and relative importance of each variable from the candidate set of models where **Δ**AICc <4, based on 739 households from 23 villages in 8 districts. Coefficient estimates are presented as contrasts from the intercept, standardised on 2 standard deviations following Gelman (2008). Where the relative importance of a variable is < 0.5, only the direction of the effect is presented. Random effects estimates of variance [standard deviation] were taken from the global model.

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	3.19 (0.74)	
Sanctuary	2.65 (1.44)	0.93
Age	-0.17 (0.24)	0.51
Awareness (1 = aware of regulations, 0 = not aware)	-	0.42
Target *jatka* (1 = target *jatka*, 0 = no *jatka*)	+	0.30
Fishing location (1 = sea, 0 = only river)	-	0.28
Compensation recipient (1 = recipient, 0 = non-recipient)	+	0.22
# of models in candidate set	31	
Random effects		
Village	2.34 [1.53]	
District	0.97 [0.99]	

Ninety-seven per cent of respondents also said that fish size was increasing from year to year, 2.8 per cent said it stayed the same, and only 0.4 per cent said it was decreasing. 44.2 per cent of respondents said that egg/fry presence was increasing from year to year, 54.2 per cent that it had stayed the same, and 0.9 per cent that it was decreasing. Modelling revealed a significant negative effect of living outside a compensation area on the probability of reporting an increase in egg presence (Table C.1b; Table C.4). When compensation area was replaced as a fixed effect by sanctuary and compensation, however, sanctuary became the most important fixed effect (relative importance 1) in top models; fishers living inside sanctuary areas were significantly more likely to report an increase in egg presence (Table 5.4; Table C.5). There was also a weak effect of fishing location (relative importance 0.37); fishers who fish in the sea were more likely to report an increase in egg presence than those who only fish in rivers. Age and awareness were less important in these models than in those for the probability of perceiving an increase in catch (relative importance 0.27 and 0.34). Plotting the BLUPs for each village within district (Fig. 5.5b) showed an effect of geography on the probability of perceiving an increase in egg presence, once fixed effects were taken into account. Households in Baushia (Barisal district) and Lalpur (Chandpur district) villages were significantly more likely to perceive an increase, whereas those in Golbunia and Nizampur (Patuakhali district), Gosherchar (Barisal), and Uttar Gobindia (Chandpur district) were significantly less likely. District was confounded by sanctuary and therefore had no effect (Table 5.4), probably due to the low variability in the dependent variable, and so the model was run again with only the village random effect to check that results did not change (Table C.1c).

When fishing households in compensation areas were asked directly about the impact of the compensation scheme on hilsa stock regeneration and catch levels, 99.6 per cent of respondents said that it has had a positive impact. Ninety-seven per cent of respondents in compensation areas also said that there has been an increase in hilsa stocks over the last five years as a direct result of the compensation scheme.

Table 5.4: Results for GLMMs of probability of reporting an increase in hilsa egg/fry presence from year to year, showing the full model-averaged coefficient estimates (standard error) and relative importance of each variable from the candidate set of models where **Δ**AICc <4, based on 739 households from 23 villages in 8 districts. Coefficient estimates are presented as contrasts from the intercept, standardised on 2 standard deviations following Gelman (2008). Where the relative importance of a variable is < 0.5, only the direction of the effect is presented. Random effects estimates of variance [standard deviation] are taken from the global model.

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	-0.13 (0.17)	
Sanctuary	1.08 (0.38)	1.00
Fishing location (1 = sea, 0 = only river)	-	0.37
Awareness (1 = aware of regulations, 0 = not aware)	-	0.34
Age	-	0.27
Target *jatka* (1 = target *jatka*, 0 = no *jatka*)	-	0.21
Compensation recipient (1 = recipient, 0 = non-recipient)	+	0.20
# of models in candidate set	19	
Random effects		
Village	0.44 [0.66]	
District	0.00 [0.00]	

(a) **(b)**

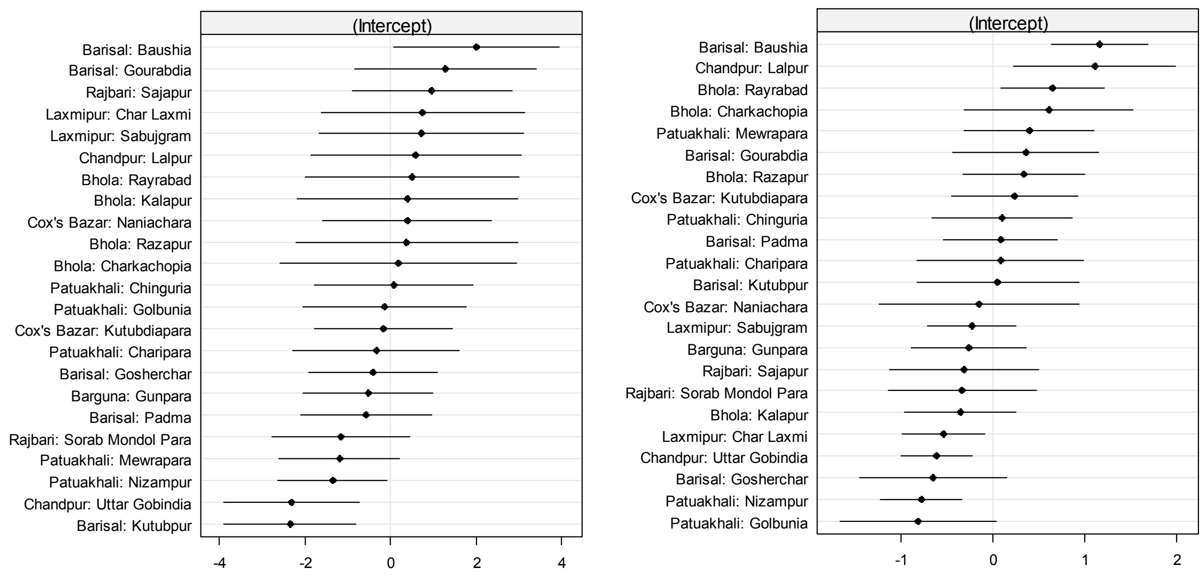


Figure 5.5: BLUPs for the village random effect. The x axes show the effect of living in a particular village in terms of **(a)** the difference in probability of reporting an increase in catch from the intercept; and **(b)** the difference in probability of perceiving an increase in eggs/fry from the intercept. Error bars show the 95% confidence interval based on the conditional variance for each random effect. Village names are prefixed by district.

5.4 Discussion

5.4.1 Potential for additionality

There is scope for individual elements of the hilsa management package to have had additionality, but the potential for overall additionality is very unclear. Evidence for the assumptions which underpin the reconstructed theory of change is often equivocal, and there are numerous uncertainties and contradictions (Table 5.5). Levels of support for the assumptions vary, but the degree to which this matters depends on its likely impact on sustainability (referred to here as importance; i.e., low importance describes a situation where even if an assumption was completely flawed, management could still be sustainable).

It is clear that overfishing is a substantive threat to hilsa stocks, although the evidence for their current status is patchy and sometimes conflicting, and they appear to be a very resilient to exploitation. It is possible that environmental factors such as pollution, damming and climate change may pose a more serious threat, but controlling fishing activities should mitigate at least some of these impacts through an increase in abundance. The validity of this assumption is therefore of moderate importance; the management approach of focusing on hilsa overfishing should still be beneficial even if other environmental factors pose substantive threats to hilsa, but this depends on the level of threat. The impact of fishing regulations may be limited, however, if environmental factors limit the availability of suitable hilsa habitat. Hilsa management is certainly flawed in terms of its focus on maximising individual species yield, which is not a sustainable approach for either the long-term food security or ecosystem health of Bangladesh, and might therefore limit the goal of socioeconomic improvement. The invalidity of the assumption that single-species management is adequate is deemed to be of high importance because projected environmental change has the potential to cause a collapse in hilsa populations, in which case the whole management strategy would need reworking. There is also evidence to indicate that focusing management on inland fisheries is not a sustainable approach although, given our limited understanding of hilsa in the marine environment, this carries great uncertainty.

The evidence for the rationale behind the protection of *jatka* is equivocal, and since multiple management actions (including the compensation scheme) are underpinned by the assumption that protection of *jatka* will maximise hilsa production, it is an important one. Although the assumption is supported by the theories of traditional fisheries science and some species-specific modelling, there is just as much evidence to the contrary. In addition, there is uncertainty surrounding the placement of the hilsa sanctuaries and the timing of the fishing bans within them. Since compensation is reportedly provided only for four months of the year and only to households within or near sanctuaries, the temporal and spatial consistency of this distribution with the availability of *jatka* is very important. Although there does appear to be some adaptive management taking place, it is clear from ecological surveys (Hasan et al. 2015) and from KIIs that at least one of the hilsa sanctuaries is no longer a suitable nursery ground, and that management is not keeping pace with environmental change. There is also some anecdotal evidence for a temporal shift in *jatka* availability. It is therefore unclear whether fishing bans for the protection of *jatka* are likely to be having an additional impact on *jatka* abundance. Because of its long length, however, the timing of the nationwide ban on *jatka* activities is perhaps less important than the timing of the sanctuary bans, and could potentially have an impact even if the timing is imperfect.

It is also clear that regulations for the protection of *jatka* (eight-month *jatka* ban, monofilament net ban, and sanctuary fishing bans) are not effectively enforced. Changes in behaviour – such as fishing at night, landing catches further inland and bribing officials – indicate that the threat of sanctions is making rule-breaking more difficult, and some fishers have been punished for breaking rules. However, there is unequivocal evidence that enforcement is nowhere near leading to 100 per cent compliance, and certainly not in a socially just manner. Fishers themselves admitted to breaking rules as a coping strategy during fishing bans, and also reported a lack of compliance amongst other fishers. Perceived and reported compliance have been related to ecological performance of MPAs in some regions, suggesting that they are strong indicators of actual compliance behaviour (Pollnac et al. 2010; Daw et al. 2011a; Bergseth et al. 2013). The perception of others not complying weakens the moral obligation to comply, ultimately leading to a breakdown of compliance and therefore ecological performance (Sutinen & Kuperan 1999).

The 15-day hilsa fishing ban for spawners seems to have stronger evidence behind it than other management actions. It has been questioned whether the ban is long enough to minimise disturbance to spawning, whether it is appropriately timed for peak spawning season, and whether it is enforced in the right areas, and so long-term monitoring – particularly to establish the level of inter-annual variability in timing of spawning – would help to resolve these questions. On the other hand, the ban appears to have a higher level of acceptance and more effective enforcement than the others, and so the assumptions surrounding this ban are probably of low importance, given that not many fishers are likely to be fishing during this time anyway. The recent extension and shift of the ban could, however, be interpreted as a socially and politically controversial statement against traditional belief, and it could be argued that compensation should be extended in turn. The rationale behind targeting spawning fish – that it should maximise recruitment and therefore production – is theoretically sound, and there is little empirical evidence to suggest otherwise. In any case, this assumption is of lower relative importance than the rationale for protecting *jatka,* because there are no other management actions based upon it.

As a package, it is possible that the rehabilitation programme (rice compensation, alternative livelihood support and awareness-raising activities) has potential to influence *jatka* fishing behaviour. The coverage of alternative livelihood support is too low for it to be having any ecological impact right now, and caution should be exercised in scaling it up as is (Wright et al. 2016). But, with the appropriate stakeholder consultation, local opinion indicates that increased coverage of a livelihood-focused intervention has more potential to incentivise compliance with rules than the compensation scheme, which does not address the issues of debt repayment or animal protein shortage during fishing bans.

There are numerous other issues that limit the opportunity for the compensation scheme to enhance compliance. There is anecdotal evidence of perverse incentives and empirical evidence of political interference in distribution, which are surely damaging perceptions of fairness and legitimacy. The scheme could thus be having some positive social impact, but without the desired influence on compliance, and therefore potential for ecological impact. Furthermore, in order for the compensation to have ecological additionality in the way that is intended, it must be conditional on compliance and, without effective enforcement, the evidence to the contrary is strong. Yet, even if the compensation allowed or prompted a very few fishers to change their behaviour, that behaviour change could eventually affect the whole population through community encouragement and peer pressure, especially if some form of community-based monitoring were to be introduced (Chapter 7). Through this process of social diffusion, in the long-term the compensation scheme could potentially be more efficient than it first appears (Goldman-Benner et al. 2012). Indeed, fishers' perceptions of the scheme were overwhelmingly positive, with 97.0-99.6 per cent reporting that they have seen a positive impact on hilsa stock and catch levels. Given the evidence above, this figure is surprising, but it could be a result of the local Government's claims, made as part of their awareness-raising activities, that numbers of hilsa are increasing. It could also be a sign of strategic bias, i.e., respondents may have been tempted to provide biased answers in the hope of receiving compensation or in support of the scheme. In future surveys more measures should be taken to reduce the risk of this bias, for example by using specialised indirect questioning techniques to estimate compliance (Nuno et al. 2013; Nuno & St John 2015), and building trust in the independence of the research team to reduce strategic responses (Bernard 2011). On the other hand, the support could be real and interpreted as evidence of social acceptance of the scheme (Bennett 2016).

Although there is an established relationship between awareness of regulations and an understanding of the rationale behind them (McClanahan et al. 2005; Read et al. 2011; Velez et al. 2014; Thomas et al. 2015), there is no useful evidence to support or refute the effect of awareness-raising activities on compliance in Bangladesh. In theory they have the potential to influence compliance, but could be limited by the level of vulnerability and therefore desperation that many fishers still experience. The impact of these activities in the future will therefore depend partially on the level of social impact that the rice compensation and alternative livelihood support have.

Table 5.5: Evidence for and against each assumption (with type of evidence), strength of support, and level of importance of this support. Level of support can be positive (strong evidence for), negative (strong evidence against), equivocal (evidence in both directions), or uncertain (no useful evidence). Importance is measured on a scale of low-medium-high, where low = if the assumption was completely flawed, management could still be sustainable.

Assumption	Evidence for	Evidence against	Level of support	Level of importance
1. *Overfishing is a substantive threat to hilsa stocks*	• Commercial CPUE, length-frequency data analysis, productivity-susceptibility analysis and systems dynamics simulation modelling for artisanal marine fishery (literature) • Inland surveys of experimental CPUE (literature) • Observed and projected human population growth (Chapter 3) • Anecdotal reports (KI fisheries scientists)	• Environmental conditions and habitat availability can have a strong influence on abundance (evidence from similar species) • Recent positive trends in fishery (literature, KIIs and fisher knowledge)	Positive	Medium
2. *Hilsa can be sustainably managed as a single species*	None	• Sustainable fisheries management requires an ecosystem approach (theory) • Climate projections indicate single-species management is unsustainable in terms of overall biomass (literature) • Morphologically similar to other species and interchangeable on market (KIIs)	Negative	High
3. *Protection of jatka maximises production*	• Systems dynamics simulation modelling in hilsa (literature) • Theories of traditional fisheries science (literature)	• Age-structured population modelling indicates that targeting juveniles is less damaging than targeting adults (Chapter 4) • Balanced fishing hypothesis (literature)	Equivocal	High
4. *Protection of spawners maximises production*	• Spawn-at-least-once principle (literature) • Systems dynamics simulation modelling in hilsa (literature) • Evidence for fish spawning aggregations as a valuable conservation focus (literature)	• Stock-recruitment relationship is complicated by environmental conditions (literature)	Positive (with uncertainty)	Low

5.	*The 15-day hilsa fishing ban coincides with peak spawning season*	• Empirical evidence for peak spawning season in correct month (literature) • (Fishers') reports	• Placement of ban may have been motivated by potential to reduce political conflict (KIIs) • Peak may be shifting (KIIs) • May not be long enough (KIIs)	Equivocal	Low
6.	*Enforcement of 15-day hilsa fishing ban is targeted to the right area*	• Important spawning grounds identified through experimental fishing (literature)	• Environmental change may have led to a shift in spawning grounds or a reduction in accessibility (KIIs)	Negative (with uncertainty)	Low
7.	*Hilsa sanctuaries hold nursery grounds*	• Areas identified through experimental fishing (literature)	• Quality of and access to sanctuary areas, one in particular, is in decline (literature and KIIs)	Negative (with uncertainty)	High
8.	*Fishing bans for the protection of jatka coincide with peak jatka abundance*	• Experimental fishing and commercial catch analysis justify timing of sanctuary fishing bans (literature)	• Temporal shift in peak *jatka* availability (KIIs)	Equivocal	High
9.	*Focus on inland fisheries management is sustainable*	• Important nursery and spawning grounds identified inland (literature) • *Jatka* is easy to target during migration downstream (literature)	• Hilsa are not strictly anadromous and so marine areas may also provide important nursery and spawning grounds (literature) • Marine fishing effort is increasing (Chapter 3)	Negative (with uncertainty)	High

10. *Regulations are effectively enforced*	• Some change in *jatka* fishing behaviour (KIIs) • DoF reports of sanctions have increased (literature) • Reports of bribes and harassment (literature and KIIs)	• Lack of resources for enforcement (literature and KIIs) • Enforcement reported as ineffective (literature and KIIs) • Perceived and reported lack of compliance with sanctuary bans (Fishers) • DoF reports of sanctions have increased (literature)	Negative	High
11. *The compensation scheme incentivises compliance*	• Level of compliance with sanctuary closures is perceived to be significantly higher for compensation recipients than non-recipients (fishers)	• Lack of conditionality on behaviour (literature, KIIs) • Perceived fairness and legitimacy of the compensation scheme is low (Chapter 6) • Perverse incentives and misallocation (KIIs and literature) • Coverage of alternative livelihood support is probably too low (literature) • Rice compensation alone is inadequate and inappropriate (KIIs and literature)	Negative	High
12. *Alternative livelihood support incentivises compliance*	None	• Low coverage (literature) • Evidence to suggest the theory is flawed (literature) • Inappropriate support (KIIs)	Negative	Low
13. *Awareness of fishery status and regulations enhances compliance*	• Empirical evidence for influence in other fisheries (literature)	• Socioeconomic situation may limit opportunity for impact (KIIs) • High level of awareness without compliance (fishers)	Uncertain	High

5.4.2 Perceptions of recent trends and management impact

The various sources used in this study can be pieced together to give some idea of recent trends in the fishery. Ecological trends, KIIs and fisher's reported perceptions all indicate that management may have had a positive impact on hilsa in terms of an increase in production and a reduction in fishing mortality, although it should also be noted that these trends cannot be attributed to any specific management action and could equally be due to environmental variation (Bennett 2016). When fishers were asked about trends in hilsa abundance and catch characteristics, both currently and in the last five years, answers were overwhelmingly positive, to the point where limited statistical analysis could be done. Although they are subjective, these data could be interpreted as complementary to recent trends in ecological data (Gaspare et al. 2015), but there is again a risk that this uniform positivity is a result of strategic bias; respondents may have provided biased answers in the hope of influencing management.

When asked about current trends in hilsa catch volume and the presence of eggs or fry, fishers living in or around sanctuaries were significantly more likely to report an increase, which indicates that sanctuary management could be having an ecological impact. The higher probability of reporting an increase in eggs or fry in and around sanctuaries is surprising, given that these sanctuaries were established to protect nursery rather than spawning grounds, and indicate that spawning grounds and nursery grounds may overlap – further evidence of complicated migratory movements. On the other hand it could be a result of awareness-raising activities (which are probably concentrated around sanctuary areas) promoting the idea that the sanctuaries are having a positive impact. The fact that compensation was not an important explanatory variable for this response provides some evidence that answers were not influenced by strategic bias in support of the compensation scheme. In the case of catch volume, older fishers were significantly less likely to perceive an increase, which could be evidence of a shifting baseline effect (Pauly 1995; Bender et al. 2013). Younger fishers might be more likely to perceive an increase because they have experienced different baseline catch volumes to those experienced by older fishers, and the specification of a time-frame might therefore have been preferable. It is unclear why those fishers who fish in the sea were more likely to report an increase in eggs or fry, but an exploration of the types of fishing gears used might provide some answers (Chapter 4).

There was also a strong spatial pattern in reported perceptions of catch characteristics. KI A2 reported frequent conflicts between small-scale and industrial fishers in coastal areas, which could be driving the district-level differences in perceptions of catch volume; industrial fishing could be limiting the impacts of management in some areas more than others. Recipients were consistently more likely to perceive an increase in both catch volume and eggs/fry in Baushia village, and consistently less likely in Uttar Gobindia and Nizampur. Nizampur (Patuakhali district) is on the Andharmanik River, where water quality is low and may no longer be suitable for hilsa (Hasan et al. 2015). The village effect could also be an indication of social desirability bias, where respondents answer questions in a manner that they think will be viewed favourably by others; i.e., some villages might be more supportive of management interventions than others for political reasons. Furthermore, it could be linked to observer effect; although no evidence of order bias was found, in three villages an additional enumerator conducted some of the questionnaires (Rayrabad and Charkachopia in Bhola district, and Baushia in Barisal district; see Appendix B.1). This inconsistency in enumerators might explain some of the village effect, particularly in the case of Baushia village, which had the most consistently different perceptions of catch characteristics. Further research would be required to interpret these perceptions at the community level.

Other limitations to interpretation of the fishers' perceptions include the fact that some of the respondents who do not actually fish themselves might not have had a reliable perception of household catch characteristics. It would have been preferable to have identified which of the respondents fish and which do not – although it is likely that most do, since women were excluded from analysis. Respondent age could have acted as a rough proxy for this, since it is probably the very young and very old males who do not fish in households who identify as fishing households. More rigorous questioning to ascertain acceptance of and compliance with management regulations would have allowed a more in-depth analysis of their potential to contribute to additionality. There is also a lack of ecological data and published studies available after 2009, other than landings data, with which to validate any of the fishers' perceptions.

5.4.3 Conclusions

This research clearly demonstrates major gaps in the ecological and social science underpinning hilsa management. The current management regime is based more on economic and normative considerations than on an understanding of hilsa life history and habitat, and there seems to be a strong political push to demonstrate impact without a clear evaluation policy or the publication of rigorous science to back it up. In particular, the lack of ecological monitoring and publication or even sharing of data is generating huge uncertainties surrounding the spatial and temporal placement of fishing bans, particularly in the hilsa sanctuaries. The application of methods to measure habitat suitability when data are limited, perhaps using a combination of remote sensing and on-site data collection, would help to identify optimal spawning and nursery areas, which could be used to re-evaluate current zoning of sanctuaries and enforcement focus (Chapter 3; Bilkovic et al. 2002).

It is impossible from this study to tease apart the potential impacts of the carrot vs. stick elements of hilsa management. Any intervention that provides payments for specific actions or behaviour should be underpinned by robust ecological and social science that links actions to additional outcomes (Bladon et al. 2014b; Chapter 2). In order to justify the continued use of the compensation scheme and alternative livelihood support as hilsa management tools, the development of a rigorous long-term social and ecological monitoring and evaluation system is required (Calvet-Mir et al. 2015; Palmer-Fry et al. 2015). This should not only enable attribution, but also an adaptive approach to management. Management strategy evaluation is recommended as a framework that would allow for adaptive management under the extreme uncertainty which hilsa fisheries present (Bentley & Stokes 2009; Butterworth et al. 2010).

This research does, however, demonstrate a need to improve enforcement of regulations, and/or to increase the prevalence of voluntary compliance through the rehabilitation activities. These management approaches so far appear to have been quite heavily shaped by the political economy of Bangladesh (Campling et al. 2012), and it is possible that the compensation scheme in particular is simply a welfare scheme that is being reframed as a conservation scheme – perhaps inappropriately. Nevertheless, the expectation of a significant improvement in the quality of top-down enforcement would be unrealistic at this stage and so, given the institutional context, the most effective way forward will be to adapt and develop the carrot-based elements of hilsa management, which may in turn reduce the requirement for top-down enforcement (Chapter 7).

More broadly, this chapter demonstrates that even when rigorous impact evaluations are not possible, useful studies can still be conducted to evaluate scope for or confidence in additionality. Further research to test empirically the causal mechanisms in the theory of change underlying hilsa management would be valuable.

Chapter 6

Does compensation for hilsa fishers reach the ‘right’ people?

6.1 Introduction

Although poverty reduction has in the past focused on minimising current shortfalls in various dimensions of wellbeing, the concept of vulnerability (i.e., the threat of future shortfalls) is now recognised as a crucial component of poverty (Ligon & Schechter 2003; Calvo 2008). Specific frameworks and terms differ across disciplines (Eakin & Luers 2006; McLaughlin & Dietz 2008), but vulnerability is generally defined as the degree to which a system or individual is susceptible to and unable to cope with the adverse effects of a stress or change (Adger 2006). Research on poverty in small-scale fisheries now emphasises vulnerability as a central dimension, particularly in developing countries (Béné et al. 2007, 2010; Béné & Friend 2011; Ferrol-Schulte et al. 2015). Targeting vulnerable fishers is thought to be as important for poverty reduction as focusing on those poorest in monetary or material-asset terms (MacFadyen & Corcoran 2002; Thorpe et al. 2007; FAO 2012).

A key determinant of vulnerability is dependence; households who are less dependent on one occupation or resource are likely to be less sensitive to, and more able to cope with, stress on an occupation or resource (Marshall et al. 2007; Junio et al. 2015; Ferrol-Schulte et al. 2015). An understanding of fishing dependence can therefore be useful in the design and effective targeting of fisheries management interventions (Jacob et al. 2010). Minimising the negative impacts of an intervention (or maximising the positive) for the most vulnerable or most dependent groups should not only enable equitable social impacts, but also improve perceptions of fairness. These perceptions can in turn promote community acceptance, compliance with regulations and thereby overall intervention effectiveness (Sutinen & Kuperan 1999; Gelcich et al. 2009; Sommerville et al. 2010a; Harrison et al. 2015).

Conservation payments are increasingly advocated as a way to meet both social and ecological objectives, particularly Payments for Ecosystem Services (PES) in developing countries (Milder et al. 2010; Clements & Milner-Gulland 2014; Ingram et al. 2014). Schemes often have an implicit or explicit side objective of poverty or vulnerability alleviation, and government-financed schemes often use measures of poverty and vulnerability as specific targeting criteria (Pagiola 2008; de Koning et al. 2011; Bremer et al. 2014). Even when schemes lack specific social objectives, it is now recognised by the conservation community that benefit distribution should always be fair and equitable (Sommerville et al. 2010a; Gross-Camp et al. 2012; Wynberg & Hauck 2014).

When interventions are specifically targeted for poverty reduction and other social objectives, benefits may still fail to reach the poorest or most vulnerable individuals (Coady et al. 2004; Domelen 2007). Unless a payment has blanket coverage, the social-ecological effectiveness and efficiency of any scheme will depend first and foremost on targeting effectiveness (i.e., to what extent the 'right' people receive payment) or, if it is voluntary, on the degree of participation by the target groups (Pagiola et al. 2005; Zeller et al. 2006; Uraguchi 2013; Poudyal et al. 2016). The literature on targeting in PES has focused largely on optimisation in terms of ecological additionality and cost-effectiveness (Wunder 2007; Kroeger 2013). Increasingly, social targeting goals are also considered (Alix-Garcia et al. 2008; Jack et al. 2009; Gauvin et al. 2010), but lessons from both conservation and development projects increasingly highlight the risk of ineffective social targeting through inclusion or exclusion errors and elite capture of benefits (Sommerville et al. 2010a; Uraguchi 2010, 2013; Pascual et al. 2014; Poudyal et al. 2016). This risk is likely to be amplified in fisheries, where governance challenges are complex and where unclear resource tenure may lead to benefit distribution issues (Wynberg & Hauck 2014; Bladon et al. 2014b). For example the Brazilian *defeso* scheme, which provides compensation to fishers for lost earnings during temporal fishery closures, may have acted as a perverse incentive and compensated free-riders who did not depend on fishing for their livelihoods in the first place (Begossi et al. 2011a; Corrêa et al. 2014).

This research focuses on the compensation scheme for hilsa fishers in Bangladesh, as previously described in Chapters 2, 3, and 5. Under the Hilsa Fisheries Management Action Plan (HFMAP), *jatka* (juvenile hilsa)[24] fishing and related activities are banned from November to July across the country. In addition, all fishing is banned in five hilsa sanctuaries for two months within this period, also for the protection of *jatka*. Hilsa fishers generally experience high levels of poverty and vulnerability (Leterme et al. 2004; Ali et al. 2010; Jentoft & Onyango 2010; Islam 2011; Deb & Haque 2011) and there are low levels of compliance with, and poor enforcement of, the fishing bans in Bangladesh (Chapter 5; Siddique 2009; Islam et al. 2016). As part of a suite of approaches for the 'rehabilitation' of *jatka* fishers, compensation is provided in the form of rice during the perceived peak period of *jatka* presence (February to May) to help fishers cope with their loss of livelihood. The primary goal of this scheme is the conservation of hilsa and associated biodiversity, but as it is funded through the national Vulnerable Group Feeding (VGF) programme, which aims to reduce food insecurity (Ahmed et al. 2009; Uraguchi 2011), it is also intended to reduce the vulnerability of affected fishers living inside and around the sanctuary areas (DoF 2002; Haldar & Ali 2014).

Since resources are limited, the compensation scheme is not open to all affected fishers, but is directed towards the poorest and most vulnerable (DoF 2002; Haldar & Ali 2014). There are no prescribed selection criteria, but the Department of Fisheries (DoF) claims to target 'real *jatka* fishers'[25], those who are 'fully dependent' on fishing for their livelihoods, and those without assets such as agricultural land or boats (M. Mome 2014, DoF, personal communication, 1st September). Each local council (see Fig. 3.2 for administrative hierarchy) is invited to put forward a list of *jatka* fishers, which is finalised through a complex process at various levels of Government (Haldar & Ali 2014). But concerns have been raised regarding political interference in the distribution of compensation, and thus its equitability (Siddique 2009; Haldar & Ali 2014). Indeed, social safety net schemes in Bangladesh, including the VGF, tend to be characterised by elite capture and high levels of inclusion and exclusion error (Matin 2000; Matin & Hulme 2003; Hossain 2007; Uraguchi 2010). A recent assessment of these *jatka* fisher

[24] Officially defined as hilsa of up to 25 cm in length (Islam et al. 2014).
[25] This meaning of this term is unclear, since it is often used in Bangladesh to describe very poor hilsa fishers, rather than strictly fishers who target *jatka*.

‘rehabilitation’ approaches, based on stakeholder perspectives, identified issues in the compensation scheme selection process and made recommendations for improvements, but shed little light on targeting effectiveness (Rahman et al. 2014a).

This chapter will therefore examine the targeting effectiveness of the compensation scheme for hilsa conservation in Bangladesh. As a *post-hoc* study, it will focus not on whether the scheme was fit for purpose at its inception, but rather on who are the current recipients. This will allow an evaluation of whether and how the targeting of the scheme could be redesigned. It will a) rank and compare relative household fishing dependence and explore the components of this dependence; b) profile the *jatka* fishers; c) investigate the correlates of compensation allocation; d) investigate the perceptions of fairness of the distribution of compensation; and e) explore how the spatial distribution of compensation would change under potential alternative targeting scenarios, asking how the strategy could be altered in order to target fishing-dependent households more effectively.

Expectations for the current distribution of compensation were made on the basis of the officially reported rationale behind the scheme (Table 6.1). Assuming that the targeting of the scheme is effective, *jatka* fishers and those with a high level of fishing dependence would be expected to be more likely to receive compensation than less dependent fishers who do not target *jatka*. It would also be expected that fishers living inside or very near sanctuary areas would be more likely to receive compensation than those living outside, due to the fact that they experience complete disruption to their livelihoods during the fishing ban, or have to travel elsewhere to fish. Households with a low income and those that had taken loans were expected to be more likely to receive compensation, since income and debt are locally-used indicators of poverty and vulnerability (Brouwer et al. 2007; Rahman et al. 2014a,b). Although assets were specifically mentioned by the DoF as characteristics for consideration, the asset measures that were available (ownership of agricultural land, boat and livestock) were incorporated into an index of fishing dependence (see Section 6.2.2) rather than being considered independently.

Household size and composition were also expected to influence compensation allocation, as relatively visible indicators of vulnerability; large households can have a lower average consumption and therefore be more vulnerable, but they can also have more members to bring into the labour force, which can reduce variance in future consumption (Lanjouw et al. 1995; McCulloch & Calandrino 2003; Christiaensen 2005). The ratio of economic earners to dependents in a household was expected to have a positive influence on receipt of compensation; a household with a high dependency ratio tends to be more sensitive to livelihood disruption and food insecurity (Christiaensen 2005). Households who consume less food, or less preferred food, as a coping strategy during ban periods were expected to suffer from food insecurity (Béné et al. 2007) and therefore be more credible candidates for compensation. Members of fisher associations were also expected to be more likely to receive compensation because fishers involved in these types of local association tend to have more influence over policy and regulations (Cinner et al. 2012; Blythe et al. 2014). Due to the subjective approach to compensation allocation and suspicions of political interference, some geographical clustering was also expected (Siddique 2009).

Table 6.1: Summary of hypothesised correlates of the probability of receiving compensation.

	Hypothesis	Explanation
Fishing dependence	Fully dependent fishers are more likely to receive compensation	The scheme is officially aimed at fully dependent fishers
Jatka fishing	*Jatka* fishers are more likely to receive compensation	The scheme is officially aimed at *jatka* fishers
Income	Low income households are more likely to receive compensation	Low income is an indicator of poverty and vulnerability (Brouwer et al. 2007)
Debt	Households who have taken loans are more likely to receive compensation	Loan taking is a typical coping strategy of the poorest fishers in Bangladesh and an indicator of vulnerability (Ali et al. 2010; Rahman et al. 2014a)
Food insecurity	Households who consume less or cheaper food as a coping strategy during ban periods are more likely to receive compensation	Households who consume less or cheaper food as a coping strategy during ban periods suffer from food insecurity, which is a dimension of poverty (Béné et al. 2007)
Household size	Larger households are more or less likely to receive compensation	Large households may be more or less vulnerable depending on the balance between production and consumption (Lanjouw et al. 1995; McCulloch & Calandrino 2003; Christiaensen 2005)
Household dependency ratio	Households with a high proportion of dependents are more likely to receive compensation	A high dependency ratio increases vulnerability (Christiaensen 2005)
Fisher association membership	Members are more likely to receive compensation	Members have more social capital and influence (Tuler 2008; Cinner et al. 2012; Blythe et al. 2014)
Sanctuary area	Households inside sanctuary areas are more likely to receive compensation	The scheme is officially aimed at fishers living inside sanctuaries because they experience a complete fishing ban and so are likely to lose the most earnings
District	Fishers in some districts may be more likely to receive compensation	Geographic clustering of targeting (Siddique 2009)
Village	Fishers in some villages may be more likely to receive compensation	Geographic clustering of targeting (Siddique 2009)

6.2 Methods

6.2.1 Data collection

800 households were interviewed between May and October 2014, by enumerators from the Bangladesh Centre for Advanced Studies (BCAS), across 19 villages in 6 districts of southern Bangladesh (Fig. 6.1; see Table D.1 for detailed breakdown). This was part of a larger survey described in detail in Appendix B.1, but here I only analyse the villages within the area in which the compensation scheme operates, both inside and outside hilsa sanctuaries (Fig. 6.2). Sections of the questionnaire relevant to this study included questions on household characteristics, fishing activities, the compensation scheme, and coping strategies (see Appendix D.1). Recent estimates of the total number of hilsa fishers in Bangladesh range from 300,000 (A. Wahab, 2015, WorldFish, personal communication, 21st March) to 500,000 (M. Mome 2015, Department of Fisheries, personal communication, 20th March) and according to M. Mome, 224,102 households received compensation in 2014, which is around 45-75 per cent of affected households, or 65 per cent according to Rahman et al. (2014a). In the study sites 60 per cent of households received compensation, and 54 per cent of surveyed households received compensation, indicating that the sample is roughly representative of the recipient and non-recipient groups (Table D.1). 632 respondents (79 per cent) were household heads and 126 respondents (15.8 per cent) were women, of whom 125 were not household heads.

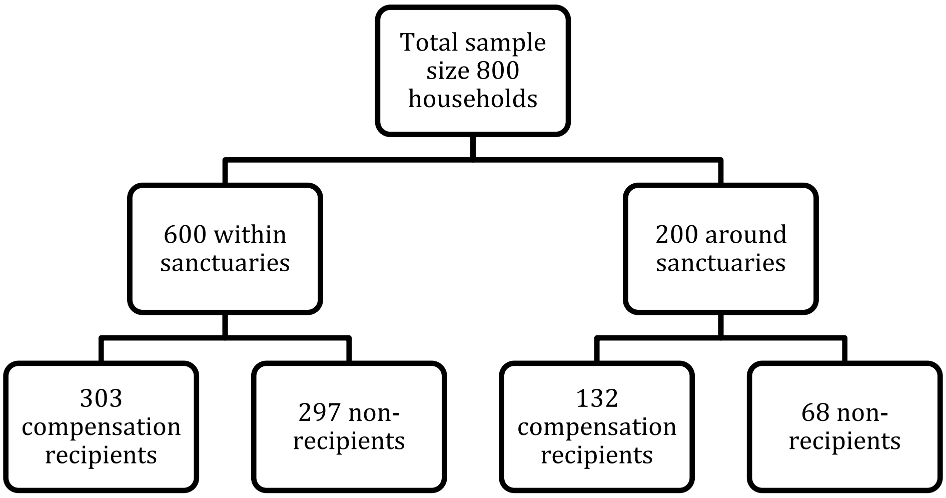


Figure 6.1: Household sampling design.

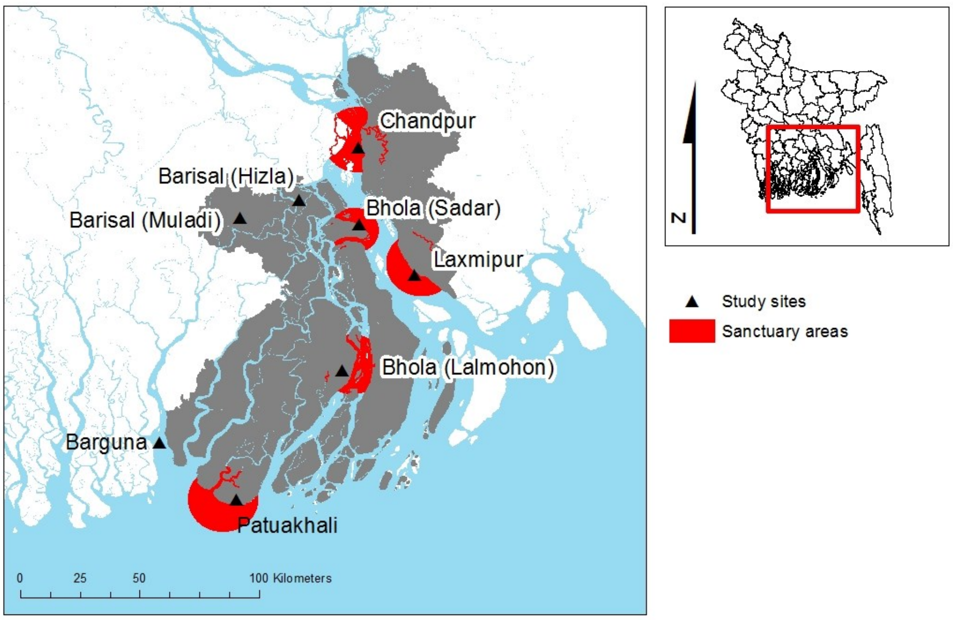


Figure 6.2: Map of study area, showing study site districts (grey) in relation to sanctuary sites (red). Each study site represents the approximate location of a cluster of surveyed villages, denoted by the relevant district name (precise village coordinates were not available). In Barisal and Bhola districts two village clusters were sampled and can be distinguished by the sub-district names (in brackets); in the other districts just one village cluster was sampled.

6.2.2 Development of an index for fishing dependence

Dependence is a multidimensional concept and therefore difficult to measure. Dependence on a resource or occupation is not necessarily reflected by level of use because there may be equally good alternatives (Marshall et al. 2007). Instead, dependence implies that there is no equivalent substitute for the resource or occupation without a loss in wellbeing. In order to explore whether there is any relationship between fishing dependence and compensation allocation, I constructed an index for fishing dependence. The development of the index was data-driven, but I used only variables which *a priori* were known to play a meaningful role in fishing dependence, based on the literature (Table 6.2). I presumed a household to be dependent on fishing when it was their primary livelihood in terms of both income and number of livelihoods; a household with high livelihood diversity may still derive the majority of their income from fishing, and a household with low livelihood diversity may derive similar levels of income from each livelihood (Hill 2011).

I used number of livelihoods together with ownership of livestock and agricultural land to develop a picture of livelihood diversity. I also expected boat owners to be more dependent on fishing; if capital is invested in fishing, this is expected to reduce adaptive capacity to move away from fishing and therefore increase dependence (Blythe et al. 2014). Gear diversity is expected to have a similar effect: although using multiple gears reduces the level of specialisation within fishing (Marshall et al. 2007; Cinner et al. 2012), it is still assumed to increase dependence on fishing in general through capital investment. Finally, I expected households who said they fish anyway as their main coping strategy during fishing bans to be more dependent on fishing than those which did not.

Table 6.2: Summary of variables used to develop an index of fishing dependence through FAMD.

Variable	Type	Description	Expected influence	Explanation
Income dependence	Percentage	The proportion of total household income which comes from fishing	+	The greater the share of fishing income in a household's total income, the more dependent the household is on fishing (Narain et al. 2008; Béné 2009; Hill 2011; Chen et al. 2014)
Other livelihoods	Binary	Households may have other livelihoods (1) or only fishing (0)	-	The more livelihood options a household has, the more able it will be to adapt to or cope with loss of fishing access (Béné 2009; Hill 2011; Cinner et al. 2012; Junio et al. 2015)
Agricultural land	Binary	Households may have agricultural land (1) or not (0)	-	Households with land may be more able to cope with loss of fishing access (Ali et al. 2010)
Livestock	Binary	Households may have livestock (1) or no livestock (0)	-	Households with livestock may be more able to cope with loss of fishing access (Ali et al. 2010)
Boat	Binary	Households may own a boat (1) or not (0)	+	It is often the fishers who have invested in the fishery who are least able to adapt to loss of access as they have sunk their capital into fishing (Blythe et al. 2014)
Gear diversity	Binary	Households may use one gear type (0) or multiple gear types (1)	+	Less highly specialised fishers, with multiple gears, are likely to be more resilient within their fishing livelihood but less able to adapt to complete loss of fishing access (Marshall et al. 2007; Cinner et al. 2012)
Illegal fishing	Binary	Fish illegally as main coping strategy during fishery closure (1) or other coping strategy (0)	+	Fishers who admit fishing illegally over other coping strategies may have few alternatives (Rahman et al. 2010; Harrison et al. 2015).

Principal component methods are commonly employed in the development of indices for multidimensional concepts, particularly wealth, to aggregate multiple variables into a unidimensional concept (Filmer & Pritchett 2001; McKenzie 2005; Zeller et al. 2006; Vyas & Kumaranayake 2006; Lalloué et al. 2013; Darling 2014). Application of Principal Component Analysis (PCA) has allowed the quantification of relative household and community vulnerability to shocks such as climate change, with implications for conservation and development policy interventions (Cutter et al. 2003; Nelson et al. 2010; Tesso et al. 2012).

Moreover, the validity of PCA in the measurement of vulnerability, resilience, and fishing dependence within fishing communities has recently been demonstrated (Marshall et al. 2007; Jacob et al. 2010, 2013). However, PCA is designed for use with continuous, normally distributed data (Kolenikov & Angeles 2009) and as such has been criticised for use in the construction of indices using discrete data (Kolenikov & Angeles 2004, 2009; Howe et al. 2008). Instead, I used factor analysis for mixed data (FAMD), a principal component method which can balance the influence of continuous and categorical variables (Pagès 2004, 2014), in the R package FactoMineR (Le et al. 2008; Husson et al. 2014). Following the methods of Vyas & Kumaranayake (2006), I carried out descriptive analyses to inform final variable selection, only selecting variables that were significantly correlated ($p < 0.05$) with the majority of the others. The index ranged from -3.44 to 2.63, so I rescaled it (-1 to +1) for more intuitive interpretation. To check for internal coherence of the index I performed an agglomerative hierarchical cluster analysis on the first dimension of the FAMD, with Euclidian distance and Ward's criterion – a preferable approach to defining arbitrary cut-off points (Lalloué et al. 2013). This differentiated households into broad groups by their level of dependence, which also allowed for more intuitive interpretation. Again, I performed the clustering in FactoMineR (Husson et al. 2014). Five households had missing data and were excluded from the analysis.

6.2.3 Data analysis

Compensation distribution

In order to identify the correlates of compensation distribution, binomial generalised linear mixed effects models (GLMMs) were fitted with the probability of receiving compensation as a binary response variable (1 = compensation recipient, 0 = non-recipient). The GLMMs were fitted as random intercept models with district and village as grouping factors in the random effects and a probit link function, in R version 3.1.2 (R Development Core Team 2014). The best random effects structures were selected using likelihood ratio tests and validation plots (Bolker et al. 2009), and models were run with Laplace approximation using the package lme4 (Bates et al. 2015). A summary and description of the explanatory variables can be found in Table 6.3. I included respondent identity (whether or not the respondent was the household head) to account for confounding variables, since respondent identity was highly correlated with age, gender, and years of education, which might in turn be expected to influence compensation allocation among household-head respondents. I also repeated the analysis using dependence cluster as an ordinal explanatory variable instead of the index, to check for consistency, but results were similar so only the analysis using the index is reported.

Collinearity among explanatory variables was explored using pairwise plots, Spearman's rank correlation coefficient, and phi coefficient. None of the variables was significantly correlated ($p > 0.05$), or at least not strongly correlated ($-0.5 > \phi$ or $r_s < 0.5$, $p < 0.05$). An information-theoretic approach to model selection was followed (Burnham & Anderson 2002; Bolker et al. 2009), as described in Chapter 5. All possible combinations of explanatory variables were fitted using Maximum Likelihood (ML) estimation procedures with the R package MuMIn (Barton 2011), and top candidate models were selected according to the corrected Akaike Information Criterion (AICc; Burnham & Anderson 2002; Bolker et al. 2009). No models were clearly superior (weights of top models were < 0.3), so those with **Δ**AICc < 4 were re-run using Restricted Maximum Likelihood (REML) estimation procedures for accurate parameter estimates (Zuur et al. 2009), which were then averaged across these models, allowing relative variable importance to be determined (Burnham & Anderson 2002; Grueber et al. 2011). Coefficients were presented for the full average, rather than the subset or conditional average, which has a tendency of biasing the values away from zero (Barton 2016). Continuous explanatory variables were standardised by two standard deviations for direct comparison of coefficients following model averaging (Gelman 2008; Grueber et al. 2011).

Models were checked for residual normality, heteroskedasticity and correlations between fixed effects and the residuals. Eight households had missing data and were excluded from analysis. To analyse spatial effects on the probability of receiving compensation, I estimated best linear unbiased predictors (BLUPs) from the global models, which measured the residual effect associated with each random effect (district and each village within district).

To explore how the spatial distribution of compensation would change under potential alternative targeting scenarios, I calculated the current proportions of households receiving compensation in each district against the proportions which would receive compensation if a) all *jatka* fishers were targeted and b) if the most fishing dependent households were targeted. Budget restrictions meant that 60 per cent of households in the study area were compensated in the year of study (Table D.1), so I selected either a) all *jatka* fishers (53 per cent of total respondents); or b) the top 60 per cent of households, in order of their fishing dependence.

Jatka fishing

Jatka fishers are reportedly the poorest and most vulnerable social group, with no other livelihoods, but previous surveys have experienced difficulty in eliciting truthful responses about *jatka* fishing (DoF 2002). Since *jatka* fishing is illegal for much of the year, fishers may not want to admit that they do it; but on the other hand, familiarity with the compensation scheme may lead fishers to exaggerate their role in *jatka* fishing. In order to gain an understanding of what socioeconomic characteristics define *jatka* fishers and to justify its inclusion as an explanatory variable in the compensation models, I explored the correlates of *jatka* fishing by fitting binomial random intercept GLMMs, following the same methods, with a logit link function. The response variable was binary (1 = household admits targeting *jatka,* 0 = household does not admit targeting *jatka*) and explanatory variables are listed in Table 6.3.

Fairness of compensation distribution

Perceived fairness is a critical determinant of acceptability in payment systems (Sommerville et al. 2010a; Clements 2012). To explore perceptions of fairness or unfairness in the distribution of compensation I fitted binomial random intercept GLMMs where the response was binary (1 = fair, 0 = not fair), and included the explanatory variables summarised in Table 6.3. I expected non-recipients, dependent fishers, low income fishers, and *jatka* fishers to be less likely to perceive compensation distribution to be fair, but I did not test the effect of *jatka* fishing due to its collinearity with dependence. I expected households with a high level of awareness of hilsa management interventions (defined as 'aware of all three interventions discussed in the questionnaire') to be more likely to report fairness, because awareness and understanding of why regulations are in place has been related to compliance, which is in turn influenced by perceptions of fairness (Agardy et al. 2011; Velez et al. 2014). Research on fairness in payment systems has found it to vary with spatial variation in governance and with local association membership (Sommerville et al. 2010a). I therefore expected fisher association members to perceive fair distribution, and variation in perceptions between villages and between districts. Again, I expected respondent identity to account for confounding variables due to its correlations with age, gender and years of schooling.

Table 6.3: List, type and description of variables investigated through GLMMs for **(a)** the probability of receiving compensation; **(b)** the probability of targeting *jatka*; and **(c)** the probability of perceiving fair compensation distribution. Blanks in expected influence columns indicate where fixed effects were not included in models.

Explanatory variables	**Type**	**Description**	**Expected influence**		
			Model (a)	**Model (b)**	**Model (c)**
Fixed effects					
Sanctuary area*	Binary	Households may live within a sanctuary (1) or outside a sanctuary (0)	+	?	
Jatka fishing***	Binary	Fishers may target *jatka* (1) or not (0)	+		
Compensation****	Binary	Households may receive compensation (1) or not (0)		-	+
Fishing dependence	Continuous	Index measuring household dependence on fishing	+	+	-
Respondent identity	Binary	Household head (1) or other (0)	?	?	?
Awareness**	Binary	Aware of all management interventions (1) or not (0)			+
Fisher association membership	Binary	Fishers may be members of associations (1) or not (0)	+	?	+
Household size*	Continuous	Number of household members	+	?	
Household dependency ratio*	Continuous	Household dependency ratio (number of economic earners/non-earners†)	+	?	
Food insecurity*	Binary	Households may use food-based coping strategies during fishing ban (1) or not (0)	+	?	
Debt*	Binary	Households may have taken a loan (1) or not (0)	+	?	
Household income	Continuous	Total monthly income per capita (BDT)	-	-	+
Random effects					
District	Categorical	6 level factor			
Village	Categorical	19 level factor			

*Not included in model **(c)**
Not included in models **(a) or **(b)**
***Not included in models **(b)** or **(c)**
****Not included in model **(a)**
†As defined in questionnaire

6.3 Results

6.3.1 Respondent profile

The respondents had a mean age of 39, household size of 5.6, household dependency ratio of 3.2 dependents per economically active household member, and 2.4 years of education. 15.8 per cent (126) of respondents were female, and only one of these was a household head. These statistics are similar to that of another recent survey of hilsa fishers carried out in the sanctuary areas (Rahman et al. 2014a) – although these authors found the household dependency ratio to be much lower at 1.49. This may be due to differences in how 'economically active' was defined (in this survey respondents were asked to state how many members of the household were earning, rather than using the number of members of working age – of which there may be those who are not working).

Seven hundred and ninety-three respondents (99.7 per cent) stated hilsa fishing to be their main income-generating activity, but this very high proportion is probably a result of strategic bias; some respondents may have overstated their involvement in hilsa fishing because of their awareness of the compensation scheme and the purpose of the survey. Local knowledge indicates that the proportion is nearer 70 per cent (B. Hossain 2015, Bangladesh Centre for Advanced Studies, personal communication, 21st January). Four hundred and twenty respondents (52.6 per cent) said that they target *jatka*, and nearly the same proportion of non-recipients to recipients said that they target *jatka* (54.1 per cent compared to 54.7 per cent). Six hundred and ten respondents (76.3 per cent) had livelihoods other than fishing, though only 8.3 per cent had more than one alternative. The main alternative to fishing was day labour, followed by agriculture (non-livestock), then business (Fig. 6.3). Seventy per cent of respondents owned homestead land, 29.8 per cent were landless, and the remainder owned agricultural land. Within this group the mean number of agricultural acres was 0.57. Households with livelihoods other than fishing had significantly more acres of agricultural land than those without

(Wilcoxon rank sum test W = 54561.5; $p < 0.05$), indicating that non-livestock farming plays a more prominent role in less fishing-dependent households. Over half (58.1 per cent) of respondents owned livestock, but ownership was not significantly associated with having other livelihoods ($\chi^2 = 0.09$; df = 1; $p = 0.77$) or with percentage income from fishing (Wilcoxon rank sum test W = 77739; $p = 0.9631$), indicating that owning livestock does not play a strong role in fishing dependence.

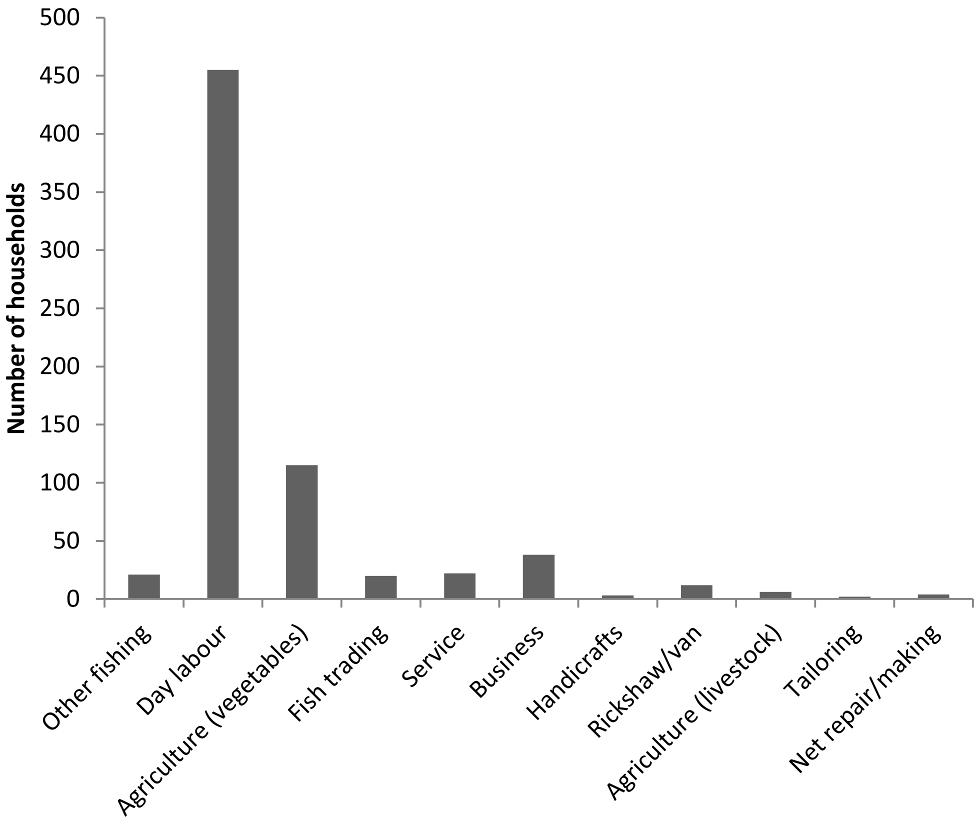


Figure 6.3: Bar plot showing the number of households who participate in various livelihoods. The total household number is greater than the total sample size because some households have more than one livelihood.

Sixty-eight per cent of respondents owned a boat and there was a significant negative association between boat ownership and having other livelihoods ($\chi^2 = 7.76$; df = 1; $p < 0.01$). As expected, boat owners also had significantly higher proportions of income from fishing (Wilcoxon rank sum test W = 48509; $p < 0.001$). Although the owners of large trawlers are likely to have other sources of income, the small and medium boat owners in this sample appeared to be constrained in their flexibility to engage in other livelihoods by their financial investment. The mean proportion of income from fishing was 82.3 per cent, showing a widespread high level of income dependency. Households without other livelihoods also had significantly higher proportions of income from fishing than their other livelihoods (Wilcoxon rank sum test $W = 109223$; $p < 0.001$). Furthermore, 40.4 per cent of respondents stated that their main coping strategy during fishing bans is to fish anyway, rather than to take another job or adopt a food-based or monetary coping strategy. It is not possible from this to conclude whether these households fish out of necessity or choice. However, since this particular response was not offered as an option by the enumerators, the high level of response supports its importance, and so its use as an indicator of dependence. Households who said they fish in ban periods had significantly higher proportions of income from fishing than others (Wilcoxon rank sum test $W = 31244.5$; $p < 0.001$) and there was a significant negative association between having other livelihoods and fishing as a coping strategy ($\chi^2 = 129.54$; $df = 1$; $p < 0.001$). The income, asset and coping strategy profiles are quite consistent with those found by Rahman et al. (2014a), although they found the proportion who fish illegally to cope during fishing bans to be much lower (27 per cent).

6.3.2 Components of fishing dependence

The first dimension of the FAMD explained 30.2 per cent of total variation. It primarily contrasts households with a high dependence on fishing (who own boats, use multiple fishing gears, fish illegally, have higher proportions of income from fishing, and have no agricultural land or other livelihoods) with households who are less dependent on fishing (who have agricultural land and other livelihoods, do not own boats, use a single gear type, do not fish illegally, and have lower proportions of income from fishing; Table 6.4). I used the first dimension as a multivariate indicator of dependence, as is the convention in the construction of socioeconomic indices (Vyas & Kumaranayake 2006; Howe et al. 2012). Although the proportion of variation explained by the first dimension is not very large, it exceeds the 30 per cent threshold necessary to avoid misclassification (Sharker et al. 2014). Furthermore it can easily be interpreted as a measure of fishing dependence, whereas the second dimension is more difficult to interpret.

Table 6.4: Variables used in FAMD to calculate a fishing dependence score for all surveyed households, with the principal component coefficients for the first three dimensions.

Variable	**Units**	**Dim 1**	**Dim 2**	**Dim 3**
Illegal fishing	1 = yes, 0 = no	1.3	-0.1	0.1
Boat ownership	1 = yes, 0 = no	0.3	0.4	-0.1
Gear diversity	1 = > 1 type , 0 = 1 type	0.5	0.2	0.7
Agricultural land	1 = yes, 0 = no	-0.6	2.2	1.0
Livestock	1 = yes, 0 = no	-0.0	0.6	-0.4
Other livelihoods	1 = yes, 0 = no	-0.6	0.1	0.1
Fishing income	BDT	0.8	-0.0	-0.1
Eigenvalue		2.1	1.3	1.0
Cumulative variance explained (%)		30.2	48.0	61.6

Hierarchical clustering defined a three-cluster solution (Fig. 6.4). Fishing dependence differed significantly between these clusters (ANOVA; η^2 = 0.86; p < 0.001) and the difference between mean dependence was highest between the medium and high clusters (Table 6.5).

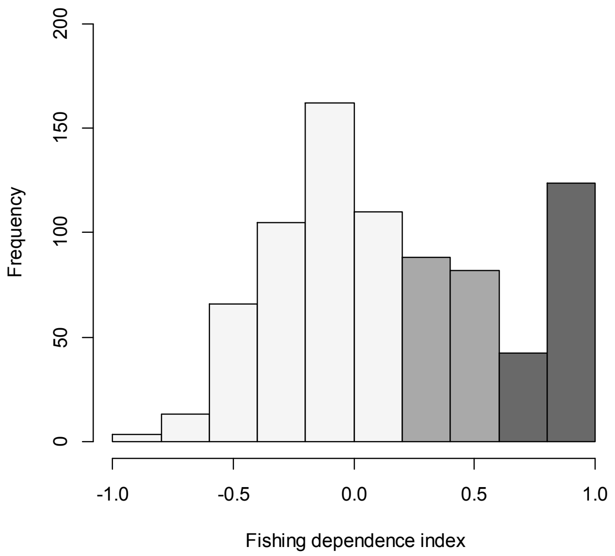


Figure 6.4: Histogram showing the distribution of fishing dependence across households, where 1 is most dependent and -1 is least dependent. The different shading differentiates groups of dependence identified through cluster analysis.

Table 6.5: Proportion of households in each cluster of fishing dependence and mean fishing dependence scores for the entire sample.

Cluster	Low dependence	Medium dependence	High dependence
Proportion of households (%)	46.4	32.6	21.0
Mean dependence score	-0.2	0.3	0.9

Table 6.6: Summary statistics and description of clusters of fishing dependence by constituent variables. The standard deviation (SD) is presented in brackets.

			Means by cluster		
	Units	**Mean (SD)**	**Low dependence**	**Medium dependence**	**High dependence**
Fishing income	BDT	82.3 (16.8)	71.7	87.6	99.7
Agricultural land	1= yes, 0 = no	0.05 (0.2)	0.1	0.1	0.1
Other livelihoods	1= yes, 0 = no	0.8 (0.4)	1.0	1.0	0.02
Boat ownership	1= yes, 0 = no	0.7 (0.5)	0.5	0.8	0.8
Multiple gear types	1 = > 1, 0 = 1	0.5 (0.5)	0.3	0.7	0.7
Illegal fishing	1= yes, 0=no	0.4 (0.5)	0.02	0.7	0.9
Livestock	1= yes, 0=no	0.6 (0.5)	0.6	0.6	0.6

A comparison of means for each constituent variable of the dependence index between clusters demonstrates internal coherence (i.e., each variable is linked to the cluster it is in; Table 6.6). All the constituent categorical variables were significantly linked to cluster (chi squared tests; $p < 0.001$), apart from livestock, which followed no clear trend and did not appear to contribute much to the index. Percentage income from fishing was also significantly linked to cluster (ANOVA; $\eta^2 = 0.52$; $p < 0.001$). There was a weak but significant negative correlation between dependence and average monthly household income (Pearson's $r = -0.17$, $p < 0.001$), which provides some validation for the use of the index as a measure of fishing dependence. Low income can increase the level of dependence on a given livelihood (Ambastha et al. 2007; Marshall et al. 2007; Narain et al. 2008).

6.3.3 Jatka fishing

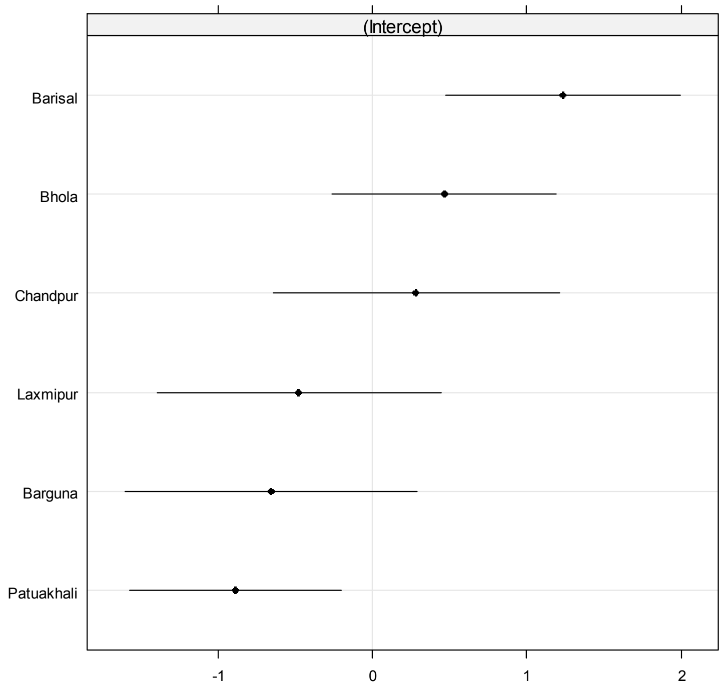
The probability of targeting *jatka* was best explained by fishing dependence, which was included in all of the top models (**Δ**AICc < 4); *jatka* fishers appeared to be marginally more dependent on fishing (Table 6.7). Household income was the next most important explanatory variable; income was lower among *jatka* fishers. The other variables investigated had very little support for inclusion in the top models (Table D.2).

Table 6.7: Result for GLMMs of probability of targeting *jatka*, showing the model-averaged coefficient estimates (standard error) and relative importance of each variable from the candidate set of models where **Δ**AICc < 4, based on 792 households from 19 villages in 6 districts. Coefficient estimates are presented as contrasts from the intercept, standardised on 2 standard deviations following Gelman (2008). Where the relative importance of a variable is < 0.5, only the direction of the effect is presented. Random effects estimates of variance [standard deviation] are taken from the global model.

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	0.10 (0.41)	
Index of fishing dependence	0.81 (0.13)	1.00
Household income (BDT)	-0.19 (0.14)	0.55
Household dependency ratio	-	0.43
Respondent identity (1 = household head, 0 = other)	+	0.26
Household size	+	0.26
Fisher association membership (1 = yes, 0 = no)	-	0.25
Sanctuary area (1 = yes, 0 = no)	-	0.23
Loan (1 = yes, 0 = no)	+	0.15
Food insecurity (1 = yes, 0 = no)	-	0.15
# of models in candidate set	77	
Random effects		
Village	0.58 [0.76]	
District	0.77 [0.88]	

The estimates for the random effects were much larger than those for the fixed effects, other than fishing dependence (Table 6.7). Plotting the BLUPs for each district (Fig. 6.5a) and for each village (Fig. 6.5b) shows that there was a significant effect of geography on the probability of targeting *jatka*. For example, households were much more likely to say that they target *jatka* in Barisal district and much less likely in Patuakhali district.

(a)



(b)

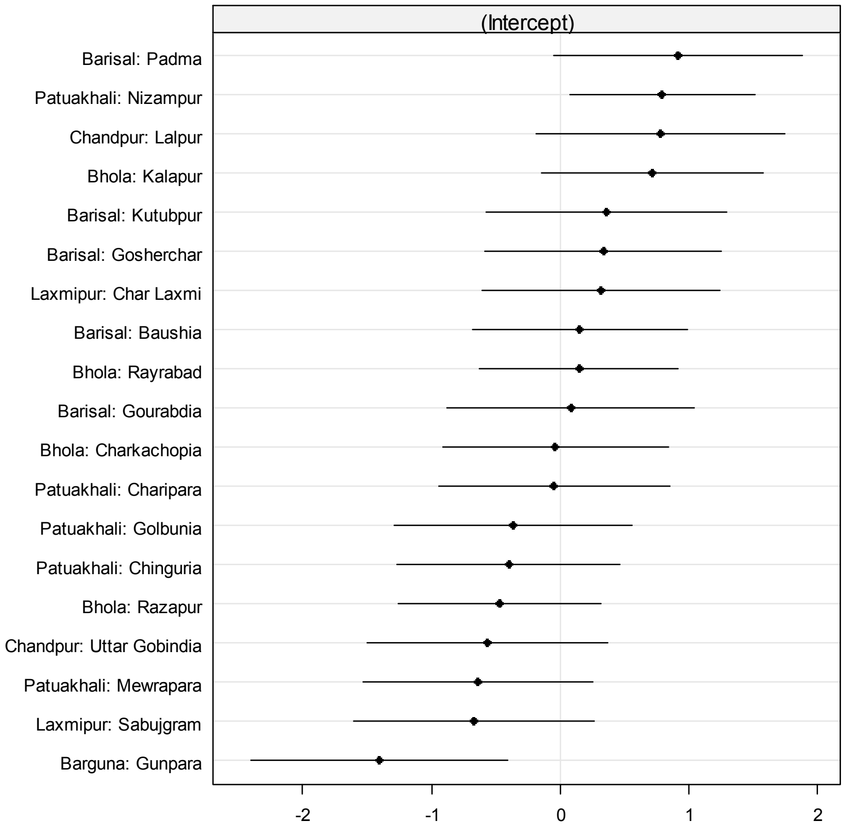


Figure 6.5: BLUPs for **(a)** the district random effect and **(b)** the village random effect. The x axes show the effect of living in a particular district or village in terms of the difference in probability of catching *jatka* from the intercept. Error bars show the 95% confidence interval based on the conditional variance for each random effect. Village names are prefixed by district.

6.3.4 Compensation allocation

Three households (0.4 per cent) who, according to recipient lists, were receiving compensation reported in the household survey that they did not receive it. The reasons given fell under the umbrella of corruption by local Government officials, a widely cited problem in Bangladesh's small-scale fisheries (Islam 2011; Haldar & Ali 2014; Rahman et al. 2014b). Another five respondents (0.6 per cent) said they received compensation even though they did not generate any income from fishing. A further two (0.3 per cent) respondents said they received compensation even though they did not fish hilsa.

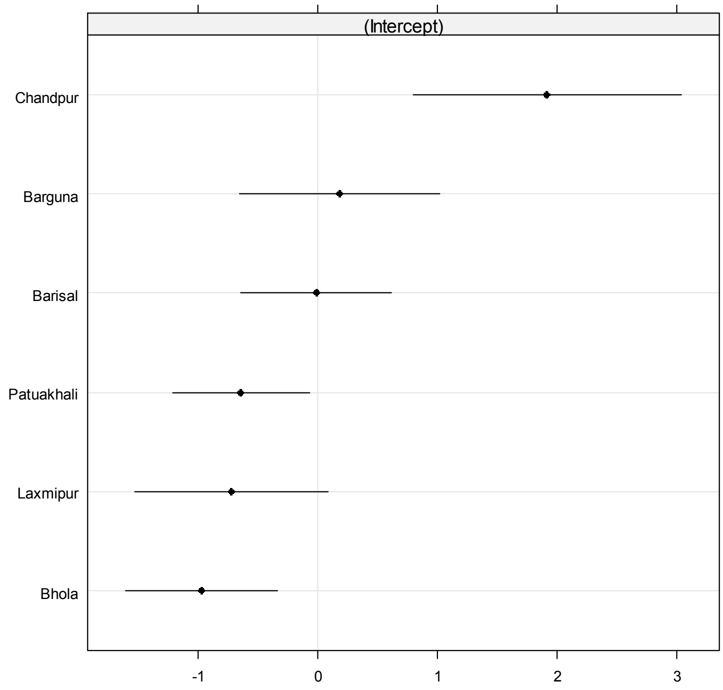
The most important fixed effect for the probability of receiving compensation was household size, which had a positive effect and a relative importance of 0.85 (larger households were more likely to receive compensation; Table 6.8), though support was weak (Table D.3). Fisher association membership also had some support for inclusion in top models (relative importance 0.68) but, contrary to expectations, households involved with fisher associations were less likely to receive compensation. Food insecurity had some support for inclusion in top models (relative importance 0.50), with a positive effect on the probability of receiving compensation, as expected. The other fixed effects had relative importance values of < 0.5 and received very little support for inclusion in the top models (Table D.3).

Table 6.8: Result for GLMMs of probability of receiving compensation, showing the model-averaged coefficient estimates (standard error) and relative importance of each variable from the candidate set of models where **Δ**AICc < 4, based on 792 households from 19 villages in 6 districts. Coefficient estimates are presented as contrasts from the intercept, standardised on 2 standard deviations following Gelman (2008). Where the relative importance of a variable is < 0.5, only the direction of the effect is presented. Random effects estimates of variance [standard deviation] were taken from the global model.

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	-0.38 (0.50)	
Household size	0.25 (0.12)	0.85
Fisher association membership (1 = yes, 0 = no)	-0.50 (0.29)	0.68
Food insecurity (1 = insecure, 0 = secure)	0.26 (0.18)	0.50
Household dependency ratio	-	0.43
Income (BDT)	+	0.41
Respondent identity (1 = household head, 0 = other)	+	0.15
Jatka fishing (1 = yes, 0 = no)	+	0.12
Index of fishing dependence	-	0.12
Loan (1 = yes, 0 = no)	-	0.12
Inside sanctuary (1 = yes, 0 = no)	-	0.12
# of models in candidate set	87	
Random effects		
Village	0.37 [0.61]	
District	1.27 [1.13]	

There was clear spatial variation in the probability of receiving compensation, and the estimates of the random effects, particularly district, were larger than any of the fixed effects. Plotting the BLUPs for each district (Fig. 6.6a) and for each village within district (Fig. 6.6b) shows the significant difference in the effect that different geographic locations have on compensation allocation, having taken the fixed effects into account. Households in Chandpur district appear to be significantly more likely to receive compensation, while households in Bhola and Patuakhali are less likely.

(a)



(b)

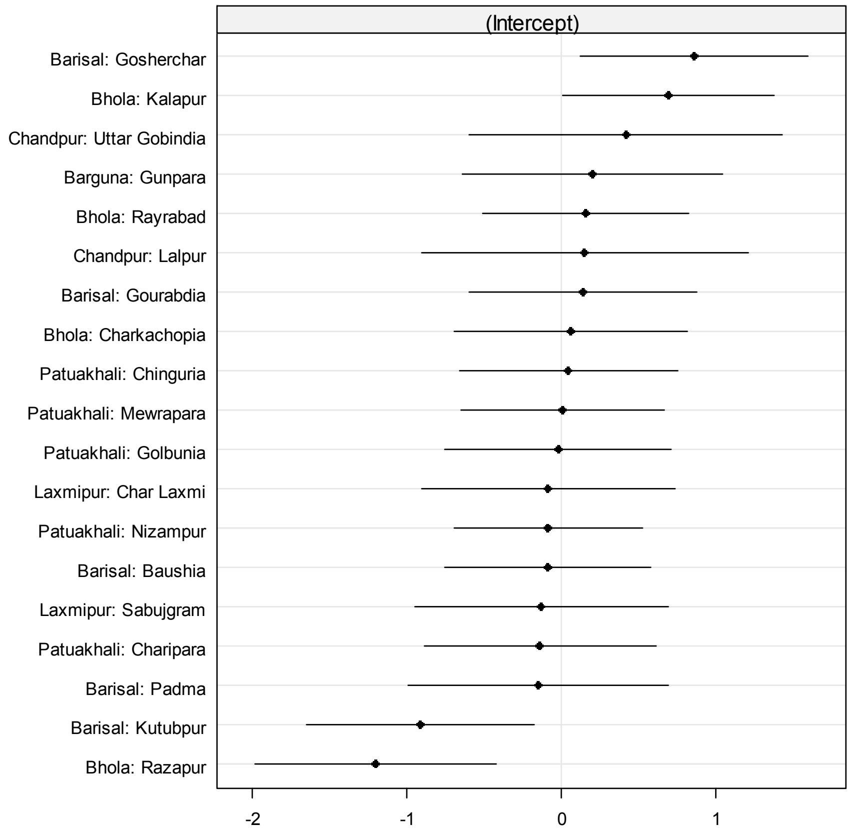


Figure 6.6: BLUPs for **(a)** the district random effect and **(b)** the village random effect. The x axes show the effect of living in a particular district or village in terms of the difference in probability of receiving compensation from the intercept. Error bars show the 95% confidence interval based on the conditional variance for each random effect. Village names are prefixed by district.

Currently, the highest coverage of fishing households by the compensation scheme is in the districts of Barisal, Barguna and, in particular, Chandpur (Fig. 6.7a). If the 60 per cent of households most dependent on fishing were targeted, coverage in Bhola, Patuakhali, Laxmipur and Barisal would increase (50 per cent, 31 per cent, 11 per cent and 5 per cent more) at the expense of Chandpur and Barguna, where coverage would drop by 65 per cent and 11 per cent respectively (Fig. 6.7b). Similarly, if only *jatka* fishers were targeted (53 per cent of households), Chandpur coverage would drop by 56 per cent and Barguna by 33 per cent, while Patuakhali coverage would increase by 44 per cent and Barisal by 28 per cent (Fig. 6.7c). This indicates that if the scheme were to be targeted more carefully according to its goals, there would be a shift in focus from Chandpur and Barguna to other districts, most noticeably to Patuakhali and Barisal.

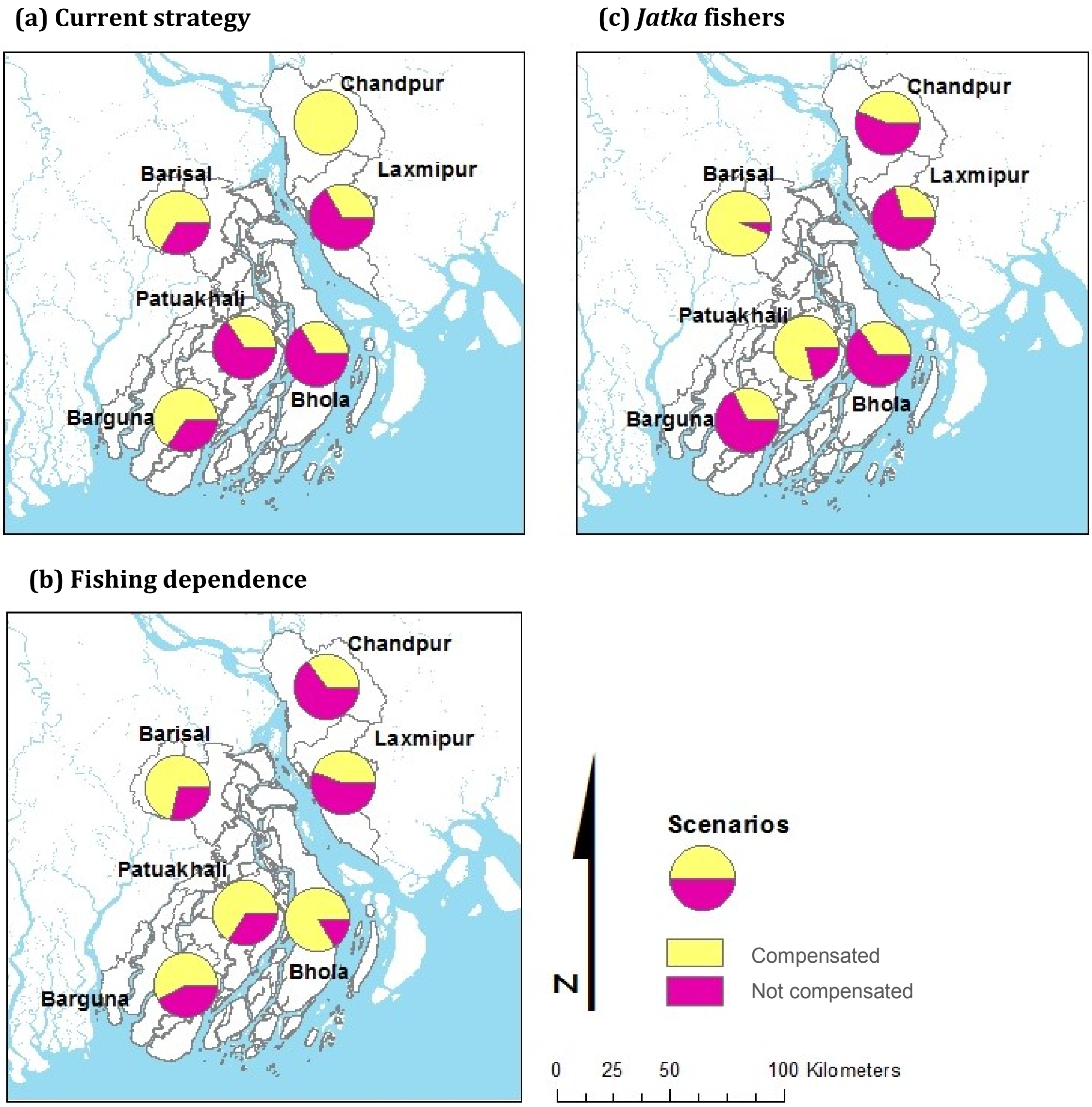


Figure 6.7: Maps showing the relative proportions of study households who would be compensated in each district under three targeting scenarios: **(a)** the current targeting scenario**; (b)** a scenario in which the most fishing dependent 60% are targeted; and **(c)** a scenario in which all *jatka* fishers (total 53%) are targeted.

6.3.5 Fairness of compensation distribution

Thirty-six per cent of respondents said they think that the distribution of compensation is fair. When asked who is currently receiving compensation, the majority of respondents said the households most dependent on fishing (99.5 per cent) and the poorest households (59.1 per cent), but 73.2 per cent also chose well-connected people and 20 per cent chose those belonging to fisher associations (Fig. 6.8) – a result which contradicts the analysis of actual compensation allocation. Most (99.7 per cent) agreed that the most dependent households should be receiving compensation, and a small remainder (0.3 per cent) said that every fisher should receive compensation.

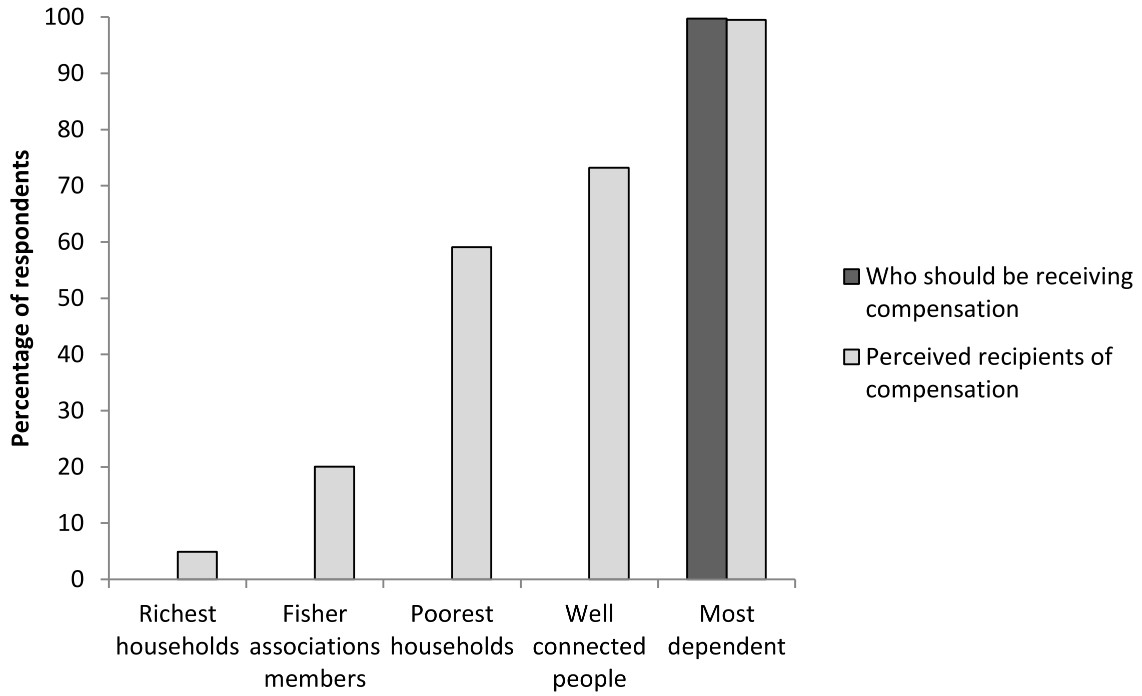


Figure 6.8: Graph showing the groups of people that the respondents (n=799) reported to be and thought should be receiving compensation. The total percentage is more than 100 because some respondents gave multiple answers.

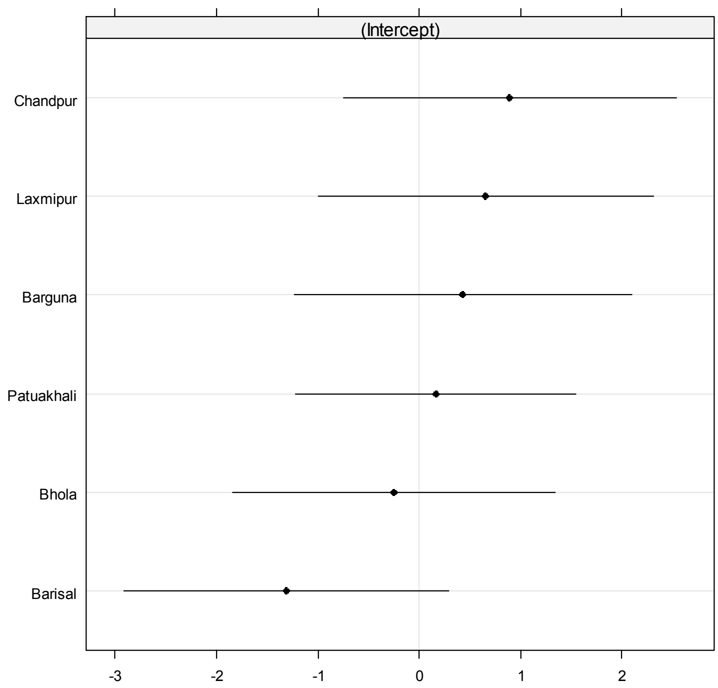
The most important fixed effect for the GLMM for perceived fairness was compensation; recipients were significantly more likely to perceive fair compensation distribution (Table 6.9). Support for the model was quite weak (Table D.4) and the standard error on this effect size was very large, but the latter was due to convergence issues caused by the absence of any non-recipients who thought the scheme was fair. Modelling also revealed a weak significant effect of fishing dependence, although this was much smaller than the random effects; less-dependent households were more likely to say the distribution of compensation was fair (Table 6.9). Plotting the BLUPs for each district (Fig. 6.9a) and each village within district (Fig. 6.9b) showed a significant effect of geography on reported fairness, once fixed effects are taken into account.

The effect of village was much stronger than that of district, where the overlap of confidence intervals with zero indicates that it is of limited importance.

Table 6.9: Result for GLMMs for the probability of perceiving fair compensation distribution, showing the model-averaged coefficient estimates (standard error) and relative importance of each variable from the candidate set of models where **Δ**AICc < 4, based on 791 households from 19 villages in 6 districts. Coefficient estimates are presented as contrasts from the intercept, standardised on 2 standard deviations following Gelman (2008). Where the relative importance of a variable is < 0.5, only the direction of the effect is presented. Random effects estimates of variance [standard deviation] were taken from the global model.

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	-11.11 (703.20)	
Compensation (1 = yes, 0 = no)	20.95 (154.09)	1.00
Index of fishing dependence	-0.40 (0.35)	0.74
Fisher association member (1 = yes, 0 = no)	+	0.26
Respondent identity (1 = household head, 0 = other)	-	0.22
Awareness (1= high, 0 = low)	+	0.21
Household income	-	0.21
# of models in candidate set	16	
Random effects		
Village	2.80 [1.68]	
District	1.09 [1.04]	

(a)



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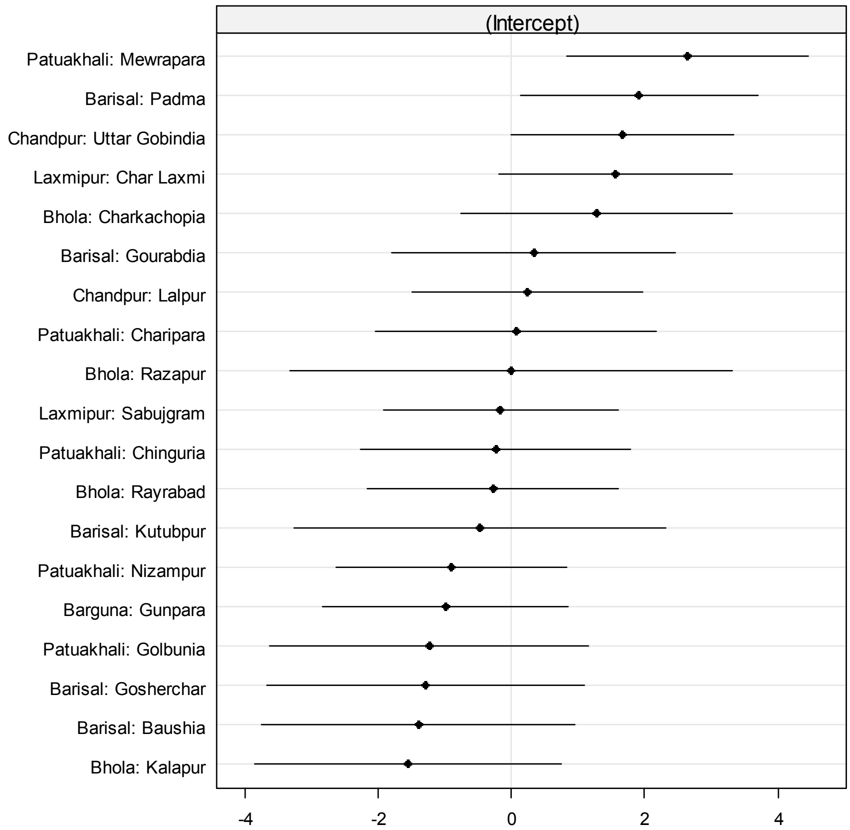

Figure 6.9: BLUPs for **(a)** the district random effect and **(b)** the village random effect. The x axes show the effect of living in a particular district or village in terms of the difference in probability of reporting fairness of compensation distribution from the intercept. Error bars show the 95% confidence interval based on the conditional variance for each random effect. Village names are prefixed by district.

6.4 Discussion

6.4.1 Compensation allocation

Currently the pattern of compensation allocation in the area under study does not strongly reflect the social goals of the compensation scheme for hilsa conservation, indicating that it is not operating as efficiently or effectively as it could be. Although the DoF claims to target *jatka* fishers and those who are completely dependent on fishing, no evidence was found to support this. There was some evidence that larger households with a higher level of food insecurity were more likely to receive compensation, which is coherent with the goals of the VGF programme, but support for these effects was weak. Contrary to expectations, involvement in fisher associations had a significant negative influence on compensation allocation. This result should be interpreted with caution since fishing households are known to be largely disorganised in south east Bangladesh (Islam 2011), overall involvement with fisher associations in the study area was low (5.8 per cent), and there is evidence to suggest that those which do exist are non-functional (Chapter 7; Rahman et al. 2014a).

The lack of clarity in the scheme's targeting criteria makes it difficult to draw any conclusions on actual levels of inclusion or exclusion error. The proportions of non-fisher and non-hilsa fisher respondents receiving compensation, and the proportions of respondents who were on the recipient list but said that they did not receive compensation, were very low. But if the error rates were to be measured using other factors in line with the rationale of the scheme (fishing dependence, income level, or *jatka* fishing), their lack of influence in statistical models indicates that the error is much higher. For example, although the scheme is aimed at *jatka* fishers, 44 per cent of non-recipients said that they target *jatka* and 56 per cent of recipients said that they do not. There is, however, a risk that strategic bias may be driving responses. It is possible that many more households did not fish, or did not fish hilsa, but may not have volunteered the information due to their awareness and understanding of the scheme. Moreover, people rarely give honest answers to questions about sensitive behaviours such as illegal resource use (Nuno & St John 2015), and so it is possible that bias was introduced by concern about the consequences of admitting to the practice of *jatka* fishing. In this case though, respondents seemed very willing to volunteer the information.

What this study did reveal is a strong spatial pattern in compensation allocation. Within the area in which the compensation scheme operates, households were significantly more or less likely to receive compensation in some districts than others. Households in Chandpur had the greatest probability of receiving compensation, once other variables were taken into account. This might be explained by the fact that Chandpur – the district where the scheme was first established – is considered to be an important landing site for hilsa and so receives a great deal of attention from the media and relatively good monitoring. It is also the site of the largest riverine nursery ground for hilsa (Mohammed & Wahab 2013), so it could be argued that by focusing on this district ecological objectives are prioritised – assuming that the scientific basis for this nursery ground is reliable (see Chapter 5). It is also possible that households in the coastal districts of Bhola and Patuakhali are less likely to receive compensation because they have less political influence (B. Hossain 2015, Bangladesh Centre for Advanced Studied, personal communication, 21st January). Due to the subjectivity of the allocation process, officials are free to use the process for political gain and the final lists are reported often to contain non-target recipients, or more than one record of one recipient (Haldar & Ali 2014; Richman & Mohammed 2016). The variation in compensation allocation between villages can probably be explained by the fact that villages with more organised and more powerful local councils have more influence in the allocation process (G. C. Haldar 2015, hilsa expert, personal communication, 30th January).

6.4.2 Alternative targeting strategies

A potential alternative strategy is to compensate all *jatka* fishers. According to the figures for the study area, this would be possible under current budget constraints, and would result in increased allocation to villages in Patuakhali and Barisal districts, with a decrease to those in Chandpur and Barguna (although of course the study villages are not necessarily reflective of the entire districts). This could be viewed as beneficial in terms of both ecological and social objectives: firstly, protection of the juvenile life stage is thought to be essential for hilsa stock recovery (although the scientific basis for this understanding may be flawed; see Chapter 4), and secondly, *jatka* fishers in this study had lower levels of household income and marginally higher levels of fishing dependence. However, support for these correlates was weak and, although local knowledge suggests that the proportions of *jatka* fishers and non-*jatka* fishers in the sample are representative (B. Hossain 2015, Bangladesh Centre for Advanced Studies, personal communication, 21st January), it is possible that the distinction between *jatka* fishers and non-*jatka* fishers was blurred by strategic bias. The challenge of eliciting true information from a respondent highlights a potential challenge in the strategy of targeting *jatka* fishers: how to identify these households. Although local councils might be able to provide accurate information on which households are very poor or depend most on fishing for their livelihoods, it may be too difficult to distinguish between those who target *jatka* and those who do not (DoF 2002). Firstly, *jatka* fishing has been described as an activity that marginal farmers and labourers switch to when income is very low, and so many *jatka* fishers are probably seasonal or occasional fishers (Haldar & Ali 2014). Secondly, although Chapter 4 indicates that *jatka* fishers are typically more likely to own boats, use a greater number of gear types, and use monofilament nets than those who do not target *jatka*, the DoF says that there is no visible distinction between *jatka* fishers and other fishers in inland areas; all artisanal fishers operating in areas where *jatka* are available will be catching them (M. Mome 2014, Department of Fisheries, personal communication, May 29th).

Targeting fishing dependence may be a more practical strategy for achieving equitable social impacts through vulnerability reduction (Béné & Friend 2011; Islam 2011; Mohammed 2011). The implementation of fishing bans disrupts patterns of access to fishing, restricting household flexibility to cope with shocks and thereby contributing to vulnerability (Marshall et al. 2007).

Dependence on hilsa fishing is universally high in and around the sanctuary areas (Rahman et al. 2014a), but this study was able to define three clusters of dependence, and households with different levels of dependence will be differentially affected by loss of fishing access. Under current budget allocation within the study areas, about 60 per cent of the most dependent households could be targeted. This would lead to large shifts in the allocation of funds, particularly from villages in Chandpur and Barguna to those in Bhola and Patuakhali. A compensation scheme that targeted dependence may also be biased towards *jatka* fishers and low-income households, since *jatka* fishers were found to be marginally more dependent on fishing and income negatively correlated with dependence.

There are some conceptual and methodological limitations to the index of fishing dependence used in this study. Due to data availability it does not represent a complete measure of fishing dependence; previous studies have used many more indicators to develop an index that explains over 70 per cent of variation – much higher than the 30 per cent in this study (Marshall et al. 2007). It would also have been preferable to also test the influence of specific non-income measures of poverty and wellbeing, incorporating dimensions such as health and housing, on compensation allocation – had they been available (Bhuiya et al. 2007).

Most of the components used to develop the index of fishing dependence in this study would be possible to verify by those administering the compensation scheme, and some of them – such as household size and boat ownership – are already noted by the DoF as factors in their selection process. One of the most influential variables in the index was the household's main coping strategy; the most dependent households tended to fish anyway as their main coping strategy during fishing bans. This would not be useful as a characteristic for the identification of compensation recipients, since it relates to illegal behaviour and so may not be willingly disclosed to the authorities. There would also be a danger here of damaging perceptions towards the scheme, through preferentially compensating those who are engaging in illegal behaviour.

6.4.3 Fairness of compensation distribution

Improved targeting effectiveness may go some way to improving perceptions of legitimacy and fairness of the scheme (Jentoft 2000; Cosens 2013; Micheli et al. 2014), which are currently poor and may be undermining the potential ecological impacts of the scheme (Pascual et al. 2014). Nearly 70 per cent of respondents reported perceptions of unfairness in current compensation distribution, which may in turn be reducing its potential to incentivise compliance with the fishing bans. Although the majority of respondents said that they think the most dependent and poorest fishers receive compensation, over 70 per cent said that the well-connected are also favoured. Procedural legitimacy (i.e., that derived from an open and transparent process of decision-making and an explanation of the choices made) is closely linked to perceptions of fairness, and both play an important role in compliance with regulations (Sutinen & Kuperan 1999). Aside from the expected association between perceiving unfairness and not receiving compensation, the strongest pattern in perceptions was between villages. This could be related to the differences in governance that can be inferred from the spatial pattern in compensation distribution (Sommerville et al. 2010a), but it could also be linked to potential observer effects, as discussed in Chapter 5. It should be noted, also, that there may always be perceptions of unfairness when there are winners and losers involved and attitudes are not always strong predictors of actual actions (Chaigneau & Daw 2015; McClanahan & Abunge 2015).

6.4.4 Conclusion

Although the complex political economy of the Bangladesh hilsa fishery cannot be ignored, this study provides evidence of the need for a) a more focused and transparent targeting strategy; and b) improved targeting effectiveness, which would require changes to the current system of administration (Chapter 7; Haldar & Ali 2014). Not only does it highlight the risk of ineffective targeting of, particularly, government-led conservation payments in developing countries

(Wunder et al. 2008), it also adds to the body of literature that recognises the need for assessments of fishing dependence, resilience, and overall vulnerability to be incorporated into fisheries management interventions (Jacob et al. 2010, 2013; Chen & Lopez-Carr 2015). An understanding of which communities and which households are likely to be most sensitive to, or adversely impacted by, new fishing regulations or future shocks plays an important role in the success of any intervention. Policy efforts to manage vulnerability tend to focus on providing new sources of resilience (i.e., reducing sensitivity to shocks) but may need a stronger focus on adaptive capacity (Tuler 2008; McClanahan et al. 2015). Following calls to improve the rehabilitation programme for *jatka* fishers (Siddique 2009; Alam 2012), the DoF have promised a future emphasis of compensation on increased provision of alternative livelihood support (Chapter 5), which is expected to contribute more to building adaptive capacity than food can. On the other hand, Ferrol-Schulte et al. (2015) argue that interventions addressing coastal livelihood vulnerability tend to be too heavily based on developing adaptive capacity and resilience, without adequately addressing exposure to risk. Although this compensation scheme is only one dimension of a broader hilsa fisheries management plan, it is worth emphasising the importance of addressing the complete range of drivers of marine resource degradation (Chapter 3).

Chapter 7

Can Conservation Trust Funds provide a sustainable framework for conservation payments in developing-world fisheries?

7.1 Introduction

Conservation interventions can only be effective if they are sustainable in the long term. Sustainability encompasses ecological, social and financial aspects, and requires institutional arrangements that enable effective governance of the social-ecological system (Orr 2002; Barrett et al. 2005; Robinson 2011). Institutions can be defined as a set of rules that constrain human agency – in a fisheries context, the rules guiding human interaction with marine and freshwater resources (North 1990; Jentoft 2004)[26]. These rules can be formal (e.g. laws, policies and regulations) or informal (e.g. social norms).

According to institutional theory, there are three types of governance structure: 1) hierarchies, which are systems of command such as a government regulation; 2) markets, which are systems of voluntary exchange; and 3) community management, which is self-regulation by communities and user groups (Vatn 2010). In practice these structures usually co-exist, and are always nested within or contain institutions at another level (Jentoft 2004). After decades of discourse around a false dichotomy between community and central government governance, it is now widely accepted that a multi-level and institutionally diverse approach to governance is most likely to lead to effective common pool resource management (Dietz et al. 2003; Jentoft 2004; Barrett et al. 2005; Muradian & Rival 2012; Muradian & Gómez-Baggethun 2013; Jones et al. 2013).

Although institutional sustainability is important for any form of conservation payment, I focus here on Payments for Ecosystem Services (PES). Not only does PES have the strongest literature base, in theory it should link ecosystem service (ES) buyers with providers through a sustainable cycle of payments (Farley & Costanza 2010). Furthermore, its requirement for conditionality and additionality means that when well designed and implemented, it should inherently be more effective in delivering conservation outcomes than other kinds of

[26] I follow Jentoft (2004) by including organisational structure in this definition. While institutions are theoretically separate from organisations, in practice a set of rules is intrinsically inseparable from the organisation that implements them.

conservation payments that lack these elements. PES requires strong institutions to structure, govern and coordinate the generation of funds from buyers; the conditional transfer of funds to providers; and also inevitable interactions with other institutions governing the broader SES, such as laws, property rights, social norms and power relations (Corbera et al. 2009; Muradian et al. 2013; Pascual et al. 2014). The conditionality of payments on actions or outcomes puts an emphasis on the requirement for monitoring and enforcement of rules, for which there must be financial and technical capacity (Chapter 2). PES is typically described by theory as a market solution, and often seen as an alternative to failed hierarchical and community governance structures, but in reality it requires state and/or community engagement and does not necessitate the involvement of markets (Engel et al. 2008; Vatn 2010; Muradian 2013; Muradian et al. 2013; Hahn et al. 2015). In developing countries with weak institutions and poor governance capacity, a hybrid institutional approach that reconciles government policies with local and international interests is likely to be most effective, and the importance of community participation has been clearly demonstrated (Fauzi & Anna 2013; Ingram et al. 2014; Kim et al. 2015).

Institutional analyses have identified various sets of enabling conditions for effective and sustainable PES – most notably trust, low transaction costs[27], and secure tenure (Brouwer et al. 2011; Escobar et al. 2013; Sattler & Matzdorf 2013; Wunder 2013; Huber-Stearns et al. 2015). However, building an institutional framework for the governance of natural resources is complicated by the fact that they are usually common pool or public goods, from which potential beneficiaries cannot easily be excluded and for which free-riding or opportunistic behaviour is likely to emerge (Muradian & Rival 2012). External intermediaries (e.g. NGOs, local organisations, consultancies or government agencies) are therefore frequently required to help establish some of these enabling conditions (Vatn 2010; Kemkes et al. 2010; Huber-Stearns et al.

[27] Transaction costs arise from the organisation or transfer of goods and services between two parties, and can be divided into three types: information costs (associated with the collection and organisation of information), coordination costs (associated with negotiating, monitoring and enforcing rules) and strategic costs (resulting from asymmetries in power and information such that some individuals or groups obtain benefits at the expense of others; Imperial & Yandle 2005).

2013; Wunder 2013; Kim et al. 2015; Schomers et al. 2015). Intermediary facilitation may often be vital in a fisheries context, where the transboundary and mobile nature of resources and stakeholders magnifies the complexity of resource governance (Bladon et al. 2014b; Chapter 2).

In addition to strong governance, PES also needs a mechanism in place to support financial sustainability. User-financed PES schemes, where environmental externalities are internalised in a closed-system loop between users and providers (the Coaesean conceptualisation of PES), should in theory be financially sustainable – particularly if they involve provisioning or supporting services (Engel et al. 2008). In government-financed schemes, financial sustainability usually depends on continued budget allocations (Wunder et al. 2008), and when payments are for the more abstract concept of biodiversity conservation, the loop between buyers and providers is less tangible. As discussed in Chapter 2, in these circumstances ongoing payments are less secure and, if payments are withdrawn, then sustainable behaviour is unlikely to continue (Swart 2003; Wunder et al. 2008; Fisher 2012) so long-term financial mechanisms need to be established (Corbera et al. 2009). For example, earmarked user fees (e.g. for the use of an MPA) or taxes (e.g. on fishing licenses) can potentially generate government revenues for long-term continuity of conservation (Pagiola 2008), or some form of buyer reserve fund (over and above that which is used for the payments) can help to finance costly monitoring and enforcement activities and thus maintain conditionality (Bladon et al. 2014b).

Some PES schemes, notably a group of watershed PES known as water funds, use a Conservation Trust Fund (CTF) model (Goldman-Benner et al. 2012), as conceptualised in Fig. 7.1. CTFs, also called Conservation Funds and Environmental Funds, are commonly defined as legally independent grant-making institutions that provide sustainable financing for biodiversity conservation and related sustainable development (Spergel & Taieb 2008). A number of authors have advocated for CTFs' potential to support economic incentive mechanisms such as PES (Spergel & Taieb 2008; Spergel & Wells 2009; RedLAC 2010a;

Goldman-Benner et al. 2012). Not only can a CTF provide sustainable financing and financial administration services for PES, it can also act as a multi-stakeholder intermediary between ES buyers and providers, facilitating collaboration, information exchange, transparency and fair governance. Experience from water funds indicates that this model can provide the extra flexibility needed for PES to operate in unconventional institutional contexts (Goldman-Benner et al. 2012).

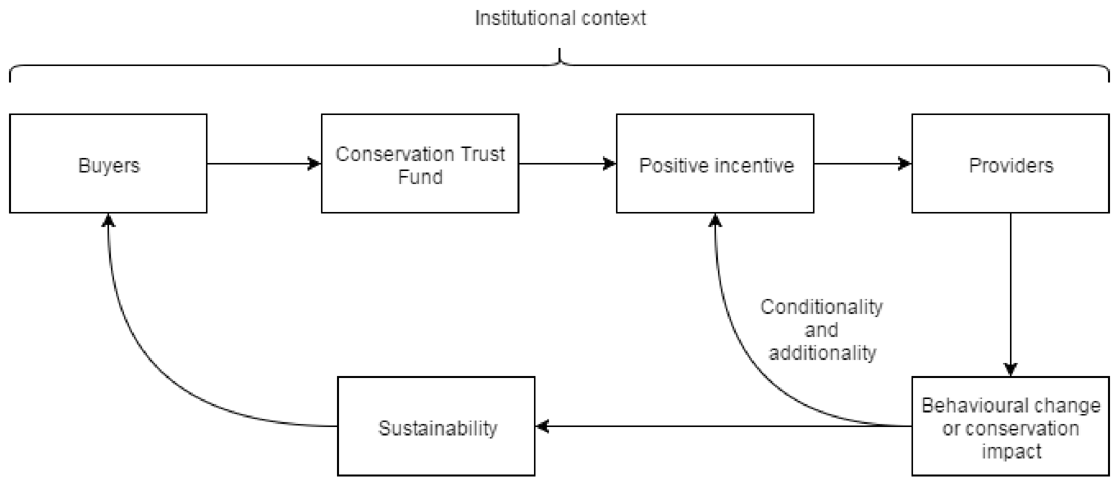

Figure 7.1: Conceptual framework for role of Conservation Trust Funds in Payments for Ecosystem Services. Payment flows from buyers to providers, conditional on behavioural change or additional conservation impact.

The broad role of CTFs in marine resource management has recently been explored (Bladon et al. 2014a). In this chapter I ask whether and how the CTF model can provide a framework for sustainable PES in developing-world fisheries, using the Bangladesh hilsa (*Tenualosa ilisha*) fishery as a case study. I begin by identifying enabling conditions for sustainable PES in developing-world fisheries, before exploring whether CTFs can support or enhance these conditions. I then assess the extent to which the current compensation scheme for hilsa fishers in Bangladesh fulfils the framework, asking how a CTF could complement, or even supersede, current institutional arrangements and thus whether it could catalyse the development of a more sustainable payment scheme.

7.2 Methods

This study combines two elements: a general exploration of the potential for CTFs to provide a sustainable framework for conservation payments in developing-world fisheries; and a specific case study of the hilsa fishery in Bangladesh, which grounds the framework in a real-world situation.

7.2.1 Case study: Bangladesh hilsa fishery

This chapter uses the Bangladesh hilsa fishery as a case study (see Chapter 3 for details) to explore the potential for a CTF to support PES in developing-world fisheries. It is a small-scale coastal marine and freshwater fishery, which is largely artisanal with a small industrial trawl subsector. Current management interventions include seasonal fishing bans for the protection of juvenile and spawning hilsa and a rehabilitation programme for fishers who target *jatka* (juvenile hilsa[28]). As part of this programme, the Bangladesh Government's Department of Fisheries (DoF) provides rice compensation to fishing households during fishing bans. The management approach has numerous weaknesses, both in concept and implementation, and it is unclear whether it actually has scope for ecological additionality (Chapter 5). Yet there is political interest in and support for moving away from a compensation scheme towards a PES, and for the creation of a hilsa CTF (Majumder et al. 2015a). It is therefore a useful case study to explore both the potential for and the challenges of implementing CTFs in developing-world fisheries.

7.2.2 Data collection

The general exploration of CTFs was based on a comprehensive literature review (published and grey literature) and key informant interviews (KIIs; i.e., interviews with individuals selected on the basis of their expert knowledge). I reviewed 12 existing CTFs, selected to give a

[28] Officially defined as hilsa of up to 25 cm in length (Islam et al. 2014).

range in terms of scale, geography, length of existence and source of finance, but with an emphasis on marine resource management (see Appendix E.1 for full list and case study summaries). I conducted semi-structured telephone interviews with 11 representatives from eight of these CTFs (representatives from the remaining four CTFs were unavailable for interview, and so these were analysed on the basis of literature alone). I also conducted interviews on the topic of CTFs with seven experts in conservation finance. All 18 interviews were conducted in November and December 2013 (Table 7.1; see Appendix E.2 for key questions).

For the Bangladesh case study analysis I used the same 36 face-to-face unstructured interviews as were used in Chapter 5, together with six additional interviews conducted at the same time (May 2014, with some conversations continuing up until January 2016; Table 7.1). Questions were focused on the sustainability of hilsa compensation scheme, Trust Funds and other public-private partnerships (PPPs) in Bangladesh, CTFs in Bangladesh, and the prospect of setting up a hilsa CTF (see Appendix E.2), but emphasis was placed on allowing the KIs to speak, only redirecting or encouraging them when they went off topic or had something particularly notable to say (Newing 2011).

Table 7.1: List and summary of key informants (KIs) for general study (11 representatives from eight Conservation Trust Funds and seven conservation finance experts) and for the Bangladesh case study (36).

	Number of KIs	Codes
General study KIs		
Mexican Nature Conservation Fund (FMCN)	2	FMCN1; FMCN2
Arannayk Foundation, Bangladesh	1	AF1
Fondo Acción, Colombia	1	FA1
Mesoamerican Reef Fund (MAR Fund)	1	MAR1
Banc d'Arguin & Coastal & Marine Biodiversity Trust Fund Limited (BACoMaB), Mauritania	2	BACoMaB1; BACoMaB2
Protected Areas Conservation Trust (PACT), Belize	1	PACT1
Caribbean Biodiversity Fund (CBF)	2	CBF1; CBF2
Phoenix Islands Protected Area (PIPA) Trust, Kiribati	1	PIPA1
Conservation finance experts	7	CF1-7
Bangladesh case study KIs		
Department of Fisheries (DoF) representatives[29]	6	D1-6
Local Government officials[4] (non-DoF)	2	L1-2
Central Government officials[30] (non-DoF)	2	C1-2
Bangladesh Fisheries Research Institute scientists	2	R1-2
Local NGO representatives	6	LN1-6
International NGO representatives	6	IN1-6
Industry stakeholders (middlemen)	5	M1-5
Industry stakeholders (members of fishers and boat owners associations)	3	F1-3
Academics working in Bangladesh	5	A1-5
Bangladeshi Trust Fund representatives	5	T1-5

[29] The DoF has representatives at each level of Government administration down to sub-district (see Chapter 3).

[30] See Chapter 3 and Appendix A for a summary of the administrative hierarchy.

7.3 Results

7.3.1 A framework for sustainable PES in developing-world fisheries

Here I identify the enabling conditions for sustainability in the context of PES in developing-world, coastal fisheries. I base this analysis on enabling conditions in the common pool resource (CPR) governance literature, adapted for watershed PES institutions by Escobar et al. (2013; Table 7.2). The diversity in social-ecological systems makes it impossible to identify a perfect institutional design for PES (Ostrom 2007), but CPR research has identified sets of conditions that promote enduring institutions, and the relevance of these in a PES context has been usefully explored (Clements et al. 2010; Fisher et al. 2010; Muradian et al. 2010). I added to and adapted the framework already developed for watershed PES (Escobar et al. 2013), with a wider review of the PES and fisheries management literature.

Table 7.2: Critical enabling conditions for the sustainability of Payments for Ecosystem Services in developing-world fisheries, adapted from Escobar et al. (2013).

Enabling conditions	References
Resource system	
• Feasibility of improving resource	Wunder (2005); Bladon et al. (2014b)
• Indicators of resource conditions	Sommerville et al. (2011); Tacconi (2012)
• Traceability of resource improvement	Sommerville et al. (2011); Tacconi (2012)
• Manageability of the system	Corbera et al. 2009; Fisher et al. (2010); Bladon et al. (2014b); Grima et al. (2016)
• Potential for profitable exploitation at a sustainable rate	Salafsky et al. (2001)
Group characteristics	
• Small resource user group	Agrawal (2001); Jack et al (2008); Fisher et al. (2010); Muradian et al. (2010)
• Prior organisational experience in resource user group	Pham et al. (2010); Wunder (2013); Mann et al. (2015)
• Appropriate leadership in resource user group	Agrawal (2001); Gutiérrez et al. (2011); Carbonetti et al. (2014)
• Social cohesion in resource user group	Gutiérrez et al. (2011); Bremer et al. (2014); Mann et al. (2015)
• Trust between stakeholders, particularly between buyers and providers	Marshall (2005); Vatn (2010); Corbera et al. (2007a); Fisher et al. (2010); Wunder (2013); Sarkki & Karjalainen (2015)
Relationship between resource and group characteristics	
• Capacity to change relationship	Pagiola et al. (2005); Sarkki & Karjalainen (2015)
• Fair and transparent benefit distribution	Pascual et al. (2014); Leimona et al. (2015); Pham et al. (2015)
• Resource tenure clarity and security	Grieber (2009); Fauzi & Anna (2013); Sattler & Matzdorf (2013); Wunder (2013); Leimona et al. (2015)
Institutional arrangements	
• Rules are simple and transparent	Pagiola et al. (2005); Read et al. (2011)
• Effective enforcement	Wunder (2005); Bladon et al. (2014b)
• Self-monitoring	Jentoft (2005); Cinner & Aswani (2007); Reynolds (2011); Sarkki & Karjalainen (2015)
• Swiftly applied, graduated, fair sanctions	Ostrom (1990); McClanahan et al. (2013)
• Low transaction costs	Muradian et al (2010); Sattler & Matzdorf (2013); Wunder (2013)
• Mechanism for sustainable financing	Corbera et al. (2009); Grima et al (2016)
External factors	
• Central government provides a supportive environment without undermining local autonomy	Vatn (2010); Huber-Stearns et al. (2013); Muradian (2013); Muradian et al. (2013); Ingram et al. (2014); Loft et al. (2015)
• Appropriate articulation with external markets	Short (2012; 2014); Fujita et al. (2012); Bos et al. (2015)

Resource characteristics

Feasibility of improvement, or the requirement for the resource (or ES) to be under a moderate level of threat (in a state of decline but not so severe that nothing can be done to reverse it), has been identified as an important precondition for the additionality of PES (Chapter 2; Wunder 2005; Escobar et al. 2013; Bladon et al. 2014b). In order to detect changes in a resource, reliable indicators of resource condition are required and, for traceability or attribution of any improvement to the payment, there must be a robust system of monitoring and evaluation in place that incorporates counterfactual scenarios (Sommerville et al. 2011; Tacconi 2012; Bladon et al. 2014b; Pham et al. 2015).

Small-scale resource systems are usually easier to manage in a sustainable way than large-scale ones, and implementation on a local or regional scale is more conducive to community participation (Corbera et al. 2009; Grima et al. 2016). External boundaries are easier to define and improvements easier to trace when systems are small (and particularly if users are in close proximity to the resource), facilitating ease of monitoring (Fisher et al. 2010; Escobar et al. 2013). In the marine environment, spatially constrained or sedentary resources are more suitable for PES than large transboundary and high-seas fisheries, but even on a small scale marine resources can be transboundary, reducing manageability (Bladon et al. 2014b).

For an effective PES, there must also be potential for profitable exploitation of the resource (or the ES it provides) at a sustainable rate into the long-term (Salafsky et al. 2001; Wunder et al. 2008). For ES providers, the benefits of the sustainable resource use that the PES incentivises, plus the payment, should exceed the benefits of exploitation in the absence of the PES; and for ES buyers, the payment must be lower than the benefits they receive from the resource, whether exploited or non-market (Fletcher et al. 2016). Without this, both potential providers and buyers would have little incentive to participate. This requirement for profitability may require bundling of ES if individual ES are unlikely to be profitably exploitable (Bladon et al. 2014b).

Group characteristics

Though of course 'small' is a relative term, small group size is generally considered to improve a group's ability to self-manage and self-monitor (Agrawal 2001). The larger the number of resource users, the greater the transaction costs are likely to be (Jack et al. 2008; Fisher et al. 2010; Muradian et al. 2010). Large and dispersed user groups are common in marine and coastal fisheries, and can be divided into smaller sub-groups for ease of management. Prior organisational experience contributes to a group's ability to coordinate monitoring and regulation (Pham et al. 2010; FAO 2011; Escobar et al. 2013; Wunder 2013; Bladon et al. 2014b). Also key to collective action is appropriate and legitimate local leadership, guided by collective interest rather than self-interest, with good communication skills and the ability to motivate individuals to coordinate actions (Agrawal 2001; Gutiérrez et al. 2011; Carbonetti et al. 2014). Social cohesion – which builds upon trust, norms and communication – has been identified as a strong determinant of success in fisheries co-management, affecting community agreement on management decisions (Gutiérrez et al. 2011). It can also determine PES outcomes through its impact on willingness to participate and comply (FAO 2011; Bremer et al. 2014; Mann et al. 2015).

Trust between stakeholders enhances institutional legitimacy, efficiency, and reduces the need for monitoring and enforcement of rules (Marshall 2005; Vatn 2010; le Coq et al. 2015), for which there is often limited capacity in developing-world fisheries (Bladon et al. 2014b). Trust between buyers and providers is particularly important, both in terms of the level to which buyers trust providers to deliver the ES and the level to which providers trust buyers to actually pay, which influences willingness to participate and comply (Corbera et al. 2007a; Fisher et al. 2010; Vatn 2010; Escobar et al. 2013; Wunder 2013). Although some studies have suggested that the involvement of intermediaries in PES reduces trust in local communities (Grima et al. 2016), usually one or more external intermediaries will be required to facilitate trust-building (Huber-Stearns et al. 2013).

Relationship between group and resource characteristics

High dependence on a resource has been identified as a condition that is important for the initiation of PES, fostering community interest and strengthening willingness to pay (Escobar et al. 2013). However, high dependence could just as well reduce the willingness of providers to participate in a PES and comply with rules, especially if they are also very poor (Pagiola et al. 2005; Sarkki & Karjalainen 2015). For sustainability, the user group should be dependent enough that they care about the resource, but not so dependent (or poor) that they have no capacity to change their relationship with the resource. On the other hand, it could be argued that, by targeting the poorest groups, PES has the potential to unite the goals of poverty alleviation and conservation in fisheries management (Chapter 6; Pagiola et al. 2005; Milder et al. 2010; Pattanayak et al. 2010; Bladon et al. 2014b). The distribution of benefits in a PES should be fair, transparent and accountable, and it should have equitable outcomes (Calvet-Mir et al. 2015; Pham et al. 2015). This fairness affects social acceptance and therefore compliance with rules (Sutinen & Kuperan 1999; Escobar et al. 2013; McClanahan & Abunge 2015), and so it is crucial for sustainability – both socially and ecologically (Pascual et al. 2014; Leimona et al. 2015).

A condition commonly cited as one of the most restrictive in PES is security and clarity of resource tenure, or resource system tenure, among providers (Greiber 2009; Fauzi & Anna 2013; Sattler & Matzdorf 2013; Wunder 2013; Leimona et al. 2015; Sarkki & Karjalainen 2015). This is crucial if providers are to be accountable for behaviour within that fishery or affecting the ES in question, and necessitates some kind of institutional framework (FAO 2011; Solazzo et al. 2015). Common pool resources and especially coastal fisheries tend to lack clear formal property rights, but some form of tenure or use rights can still be established through collective action, keeping transaction costs to a minimum (Clements et al. 2010). These use rights can be based on customary rights, where appropriate, and be made conditional on sustainable management in the PES system (Suyanto et al. 2007; Fauzi & Anna 2013; Kerr et al. 2014).

Institutional arrangements

The first step in establishing effective institutional arrangements for natural resource management is to set and enforce rules (Barrett et al. 2005). Enforcement of the rules requires monitoring for compliance and a sanctioning mechanism which, if applied, should include careful consideration of why rules have been broken. The sanctioning mechanism should be swiftly applied and graduated (i.e., the punishment should fit the crime and allow the offender to re-engage), factors that influence perceptions of fairness (Ostrom 1990; McClanahan et al. 2013; Escobar et al. 2013). In PES there should be enforceable conditionality on clear behaviours or outcomes that are achievable by the providers (Wunder 2005). However, a common limitation in developing-world fisheries is a lack of capacity for this enforcement (Bladon et al. 2014b). In these circumstances, hybrid institutions for collaborative management can facilitate self-monitoring (Jentoft 2005; Cinner & Aswani 2007; Sarkki & Karjalainen 2015). Transparency and simplicity of rules improve ease of implementation, avoid unintentional or uninformed non-compliance, and reduce barriers to participation (Pagiola et al. 2005; Read et al. 2011; Escobar et al. 2013). Local participation in design can contribute to this transparency, and enhance durability (Agrawal 2001; Fisher et al. 2010; Muradian et al. 2010; Reynolds 2011; DeCaro & Stokes 2013; Pascual et al. 2014). In a public PES scheme, transparency should be monitored by an independent authority, in order to avoid potential corruption or mismanagement of resources (FAO 2011).

Transaction costs play an important role in determining the sustainability of PES, and can be prohibitively high (Sattler & Matzdorf 2013; Wunder 2013). Intermediaries (particularly buyer and provider institutions), are usually required to help reduce these costs, especially when schemes are small-scale (Muradian et al. 2010; Wunder 2013). Sustainable financing should be available for a period of at least 10-20 years (Corbera et al. 2009; Grima et al. 2016). Given the fact that PES rarely conforms to its Coasean conceptualisation, some mechanism for this sustainable financing needs to be established, whether to cover operational costs or as insurance should buyers withdraw.

External factors

Pre-existing institutions and policies can enhance or impede sustainable PES (Corbera et al. 2009; Huber-Stearns et al. 2015; Mann et al. 2015). It is widely recognised that resource users should have some autonomy from central government to determine rules through formal or informal mechanisms (Ostrom 1990; Escobar et al. 2013). Government-financed and user-financed PES alike need a hybrid and multi-level form of governance in which providers have rights to organise (Vatn 2010; Huber-Stearns et al. 2013; Muradian 2013; Muradian et al. 2013; Ingram et al. 2014; Loft et al. 2015). Centralised interventions often fail to enable all local ES providers to participate (Clements et al. 2010). If PES is to facilitate truly adaptive and sustainable management, these providers need the power, capacity and flexibility to make and modify rules (Hayes et al. 2015; Mann et al. 2015). There is also an abundance of more general evidence to suggest that this kind of hybrid governance structure strengthens small-scale, developing-world fisheries management (Imperial & Yandle 2005; Cinner & Aswani 2007; McClanahan et al. 2009; McClanahan et al. 2013; Gianelli et al. 2015).

High articulation with external markets (i.e., where there is an external market for the resource, to which providers are well connected) is typically understood to have a detrimental impact on resource use, but the potential effects of competition on prices can also have positive impacts on willingness to participate in PES or comply with rules (Jack et al. 2008; Escobar et al. 2013), and so PES is sometimes articulated with external markets (Gómez-Baggethun et al. 2010; Boisvert et al. 2013). For instance, price premiums and increased price stability may attract a fisher to participate in a PES, and so where PES promotes sustainable fishing practices, articulation with the external markets that are offering these premiums can be beneficial for sustainability (Short 2012, 2014; Fujita et al. 2012; Bos et al. 2015).

7.3.2 Can Conservation Trust Funds support sustainability in PES?

Although CTFs vary in purpose, legal and political context, human resource capacity, and donor requirements, here I define a CTF as any institution that: a) is governed by an independent board (of directors, trustees or otherwise) with stakeholder representation; b) mobilises (and often invests) funds and keeps them separate from other sources of finance; and c) re-grants these funds for designated conservation activities (Bladon et al. 2014a). Although some CTFs are becoming more directly involved in the implementation of these activities (KI CF2), they can be distinguished from other kinds of conservation NGOs by their role as a 'bridge' between donors and implementing organisations. The first CTFs emerged in the early 1990s as mechanisms to absorb and disburse the capital that was becoming available through bilateral debt-for-nature swaps, whereby foreign debt owed by a developing nation is forgiven in exchange for local investments in conservation (Bayon & Deere 1998). More than 70 CTFs have since been established to finance conservation at national and regional levels, with a concentration in Latin America and the Caribbean, many of which support marine conservation objectives (see Table 7.3). For example, a KI from the Latin American and Caribbean Network of Environmental Funds (RedLAC; KI CF2) reported that of the 22 member CTFs, 64 per cent support marine conservation – particularly marine protected areas (MPAs).

CTFs have the potential to advance PES initiatives in a number of ways (see Table 7.4), although it is important to note that – as with any institution – this potential can only be fulfilled if best practice is followed (Spergel & Mikitin 2013; Bladon et al. 2014a). A key advantage is the effective financial model that they can provide (Spergel & Taieb 2008; RedLAC 2010b; Goldman-Benner et al. 2012). CTFs often bring together a catalytic range of donors and streams of financing in a manner that is rare in conservation, providing a reliable flow of funds that is not subject to the vagaries of project funding and can be consistent with the absorptive capacity of grantees (Klarer & Galindo 2012; TNC 2012). Funds can be structured as endowments, sinking funds, revolving funds, or any combination thereof (Table 7.5). Investment strategies tend to be conservative, minimising risk while focusing on capital growth; revenue is usually divided into that to be withdrawn and spent on operations and that which is reinvested, providing insulation from annual variations in revenue flows (Norris 2000; Spergel & Taieb 2008; Grafton et al. 2008; Mathias & Victurine 2012). Nevertheless, KIs pointed out that initial capitalisation of CTFs can be difficult (KIs FMCN1; MAR1; CBF1; CBF2; CF3; PIPA1; e.g. Case Studies 5 and 6, Appendix E.1), and that donors are in general becoming less interested in giving endowments, unless there is a great deal of leverage associated with the donation – for example through political co-financing commitments (CF5; CF7; e.g. Case Study 11, Appendix E.1). For this reason, more and more CTFs are incorporating alternative financing mechanisms such as PES, user fees, taxes, REDD+, biodiversity offsets, and environmental compensation, which generate revolving funds on a continuous basis (CF1; CF2; CF5; CF6; Spergel & Taieb 2008; RedLAC 2010a). For instance, a major source of revenue for Belize's Protected Area Conservation Trust (PACT) has been an earmarked tourism tax, while Mauritania's Banc d'Arguin Coastal and Marine Biodiversity Trust Fund (BACoMaB) was partially capitalised with finances from the nation's EU fisheries agreement (see Case Studies 11 and 4, Appendix E.1). When a CTF acts as a trustee or administrator of PES revenues, a diversified financial structure should mean that it can provide a sustainable source of financing even if one or more buyers were to withdraw (e.g. Case Study 4, Appendix E.1; Goldman-Benner et al. 2012). Furthermore, it may be able to offer microcredit, grants or non-financial incentives to cover otherwise prohibitive PES start-up costs and encourage PES development, especially if the CTF is mature (RedLAC 2010b). Throughout the operational phase it could also directly reduce transaction costs by helping to finance costly monitoring and enforcement activities (Spergel & Taieb 2008). On the other hand, operational costs are a challenge for many CTFs (FMCN1; FMCN2; BACoMaB1; MAR1; CF1) and keeping these costs low requires the same careful financial analysis entailed by the planning of any conservation instrument (Norris 2000; Bos et al. 2015).

Table 7.3: Examples of Conservation Trust Funds (CTFs) that support marine conservation objectives. This table is based on full Case Study summaries in Appendix E.

CTF	Country / region	Year est.	Institutional structure	Fund generation	Fund delivery
Phoenix Islands Protected Area (PIPA) Trust	Republic of Kiribati	2009	NGO governed by a 3-member board under the laws of Kiribati, with 2 permanent staff	USD 5 million PIPA Trust Endowment Fund capitalised by Conservation International and Kiribati Government and some sinking funds from multilateral aid and regional governments	Funds allocated to PIPA and national Government for MPA management
Banc d'Arguin Coastal and Marine Biodiversity Trust Fund (BACoMaB)	Mauritania	2009	Foundation governed by a 7-member board under UK law, with 2 permanent staff	€10 million endowment capitalised through bilateral aid, philanthropic foundations & EU Fishing Agreement	Funds disbursed for MPA management but plan to extend support to oceanographic institute and communities in MPAs
Mesoamerican Reef (MAR) Fund	MAR ecoregion (Honduras, Belize, Mexico, Guatemala)	2009	Charitable foundation governed by a 12-member board (can be up to 17 members) under US law, with 9 permanent staff	USD 23 million endowment	Funds disbursed to national NGOs, community institutions, governmental and scientific organisations for marine and coastal ecosystems of and watersheds draining into MAR
Caribbean Biodiversity Fund	Caribbean	2012	UK charity governed by a 2-member board, Secretariat based in Bahamas	USD 42 million endowment capitalised by GEF, TNC and KfW.	Funds disbursed to national level CTFs for marine and terrestrial PAs in the region
The Protected Areas Conservation Trust (PACT), Belize	Belize	1996	Foundation governed by an 11-member board under Belize law with 17 staff	Revolving fund financed through a conservation tax, a BZUSD 6 million endowment and a sinking fund financed through a debt swap	Funds disbursed to national public institutions, municipalities and NGOs for national PA network
Mexican Fund for the Conservation of Nature (FMCN)	Mexico	1994	Civil association governed by a 19-member board and 32-member general assembly with 44 staff	USD 120 million endowment capitalised by US and Mexican Governments and sinking funds from various sources including philanthropic donors. USD 76 million of this is managed separately as the Fund for Natural Protected Areas (FANP), segregated from other programmes	Funds disbursed to NGOs, organised rural communities and conservation professionals for national PAs, watersheds and forests, oceans and coasts and 'special programs'

Table 7.4: Summary of whether a best practice Conservation Trust Fund (CTF) could help to catalyse the improvement (if already fulfilled) or development (if not yet fulfilled) of critical enabling conditions for the sustainability of Payments for Ecosystem Services.

Enabling conditions	Potential for CTF to support condition
Resource system	
Feasibility of improving resource	×
Indicators of resource conditions	✓
Traceability of resource improvement	✓
Manageability of the system	×
Potential for profitable exploitation at a sustainable rate	×
Group characteristics	
Small resource user group	✓
Prior organisational experience within resource user group	✓
Appropriate leadership within resource user group	✓
Social cohesion within resource user group	✓
Trust between stakeholders, particularly between buyers and providers	✓
Relationship between resource and group characteristics	
Capacity to change relationship	✓
Fair and transparent benefit distribution	✓
Resource tenure clarity and security	✓
Institutional arrangements	
Rules are simple and transparent	✓
Effective enforcement	✓
Self-monitoring	✓
Swiftly applied, graduated, fair sanctions	✓
Low transaction costs	✓
Mechanism for sustainable financing	✓
External factors	
Central government provides a supportive environment without undermining local autonomy	✓
Appropriate articulation with external markets	✓

Table 7.5: Summary of types of fund and their advantages and disadvantages. It is common for the financial structure of a CTF to combine these features. Table adapted from Bladon et al. (2014a).

	Advantages	Disadvantages
Endowment fund Where the financial assets of a fund are invested and only the income from this investment is used to finance activities	• Suitable for PA and national park financing, which require a long-term source of financing. • Can cover a CTF's basic operational costs • Can be useful to leverage additional sources of funding	• Ties up substantial amounts of resources with relatively low returns • Take time to start producing income • Least attractive to donors
Sinking fund Where activities are financed using both principle and investment income over a fixed period of time (usually 6-15 years), or until the fund sinks to zero	• Suitable when large amounts of money are required on a one-time basis • More attractive to donors as they like to see the effects of money being spent	• Lack of permanence
Revolving fund Where a fund is replenished or augmented on a continuous basis, for example through earmarked taxes, user fees or PES	• Can last in perpetuity if the source is financially sustainable • Can cover a CTF's basic operational costs • Can connect ES beneficiaries with providers	• If used in isolation, withdrawal of the source of replenishment could cause the collapse of the fund

Through coordination with, or even involvement of, the corporate sector, a CTF could also help to articulate PES with external markets, where appropriate. Research suggests that the seafood sector is a potentially significant source of investment for PES, which could support the transition to sustainable fisheries (Vallejo et al. 2009; Short 2012; Blasiak et al. 2014; Micheli et al. 2014; Chaplin-Kramer et al. 2015). Corporate sector partnerships are becoming increasingly common among CTFs (see Case Studies 5, 10, 11 and 12, Appendix E.1; CF1; CF6); for instance, Ecuador's FONAG has received funds for PES from a water-bottling company, a brewery and an electrical utility (Case Study 11, Appendix E.1). By increasing access to premium seafood markets that are potentially more interested in the full provenance story of relevant products than standard markets, a CTF could generate funds for PES that are more likely to have an additional social impact and improve accountability for sustainability within the seafood supply chain. This would be particularly valuable for developing-world fisheries, most of which are currently financially limited from gaining access to these markets (Blackmore et al. 2015).

But a CTF is more than a conduit for funding. Best practice standards direct that a CTF should be governed by an independent and participatory board (Spergel & Mikitin 2013). Board members are usually from diverse sectors – national and international policy-makers, organisations capable of assisting with capacity-building, grantees, NGOs, other CTFs and institutions with financial expertise – and so CTFs are often referred to as PPPs. In theory, this governance structure positions CTFs to play an important intermediary role between ES buyers and providers (e.g. Case Studies 3 and 10, Appendix E.1), in which they can help to identify potential buyers and to achieve economies of scale by negotiating agreements between multiple buyers and providers, or for the bundling of multiple ES (CF2; CF4; Spergel & Taieb 2008). This brokerage role is particularly vital in remote developing-country areas where providers can be disparate and lacking social cohesion. The independent nature of a CTF should also allow it to create the dialogue and build the trust that is required between buyers, providers and other interest groups, which often have divergent interests (le Coq et al. 2015). Although other intermediaries – particularly local institutions – can also play this trust-building role (Corbera et al. 2007b; Petheram & Campbell 2010), their relationships can be complex and competition may arise (Pham et al. 2010). Furthermore, local institutions have limited power in countries where they are strongly influenced by government, as Pham et al. (2010) observed in Vietnam.

Whether user- or government-financed, PES contracts can restrict decision-making rights and fail to engage providers in their design and implementation (Hayes et al. 2015). Through independence of the board from any single organisation, CTF governance should remove the opportunity for central government to overrule or undermine local management decisions in a PES, and under best practice there should be resource-user representation. The governance structure also facilitates permanence; a PES administered by a CTF is more likely to withstand changes in both local and central government and political priorities than a state-managed mechanism (see Case Study 1, Appendix E.1), and is more likely to persist than a PES that is directly driven by the priorities of an NGO. Although this independent and participatory governance might also allow for more adaptation of priorities and strategies over time than a state-managed scheme would, flexibility ultimately depends on the length of the PES contract. The potential for a CTF to be established in an offshore location could be advantageous for PES in countries with political or financial instabilities (e.g. Case Study 5, Appendix E.1), lack of transparency and/or confidence in the country's governance (e.g. Case Study 6, Appendix E.1), and no laws allowing for tax exemption (e.g. Case Study 4, Appendix E.1; CF2; Norris 2000; Klug et al. 2003; Spergel & Taieb 2008).

Although its governance structure is, on paper, a great strength of the CTF model, many KIs (CF2; CF5; CF7; FMCN1; BACoMaB2; MAR1) said that the development of an effective governance structure is a key challenge in the creation of a CTF (e.g. Case Study 4, Appendix E.1). All CTF representative KIs also noted that it is still essential to establish government support for a CTF at an early stage – a requirement shared with PES – and that this can also be a challenge. Not only is it essential for there to be national support and acceptance of the CTF, but government backing can increase funding opportunities and, in the case of multilateral donors such as the World Bank, it is often a prerequisite for involvement. This government buy-in is often achieved by choosing a high-level government representative for the board (see Case Studies 1, 3 and 11, Appendix E.1). But to an extent, said KI FMCN1, 'the political will is either there or not' (e.g. in Case Study 6, Appendix E.1, support was there from the start). If a CTF has successfully garnered political buy-in, it may pave the way for a PES, but if the timing is wrong and the government is not receptive, then a CTF may be no more able than any other institution to catalyse support for a PES. Political support should also be approached with caution; a CTF needs the commitment and support of people without strong political agendas. This requirement was demonstrated by the failure of the Yasuni-ITT Trust Fund, in which the ownership of the President of Ecuador and the suspect integrity of his administration is likely to have played a fundamental role in its demise (Case Study 9, Appendix E.1). Furthermore, too much government involvement can limit funding opportunities due to perceived risk of fund diversion (Case Study 11, Appendix E.1).

CTFs have potential to advance PES through their role as grant-making institutions. CTF procedures tend to be less complex than those of government and, as a result, a CTF may be able to disburse funds more efficiently and more according to needs (e.g. Case Study 2, Appendix E.1). Best practice CTFs have a clear vision and strategy for grant-making in place from the outset, established through a consultative process that takes into account the relevant political, legal and governance contexts, and all potential sources of funding (CF2; FMCN1; Spergel & Mikitin 2013). Some CTFs are established to support national environmental plans or strategies (e.g. Case Study 8, Appendix E.1), some to support the management of PAs or PA networks (e.g. Case Studies 3, 4, 5, and 11, Appendix E.1), and others to fulfil broader conservation missions (e.g. Case Study 1, Appendix E.1). Within its broad strategic plan, a CTF could make grants to projects that either create or profit from a PES, putting out calls for proposals with defined objectives. Funds can be granted to any capable implementing organisation (government agencies, NGOs or community organisations) through an open and competitive process of the CTF soliciting and evaluating implementation proposals based on transparent criteria (Spergel & Taieb 2008). This would contribute to trust in the fairness of the resultant PES programme, which should in turn help to attract additional funding from private and international donors who may not otherwise have been interested buyers, particularly in countries with unstable political systems (e.g. Case Study 6, Appendix E.1). However, some CTFs (particularly the older ones) have suffered from lack of a clear grant-making strategy, which can result in the CTF being inundated with proposals and acceptances made on the basis of a 'recommendation' rather than a transparent process, leading to mistrust in the fairness of the process (e.g. Case Studies 2 and 8, Appendix E.1; CF2; CF3).

CTFs are evolving from being primarily grant-making institutions to taking a more active role as facilitators and policy advocates (CF2; Spergel & Taieb 2008). For example, Belize's PACT has divided its grant-making strategy in project grants and capacity-building grants (Case Study 11, Appendix E.1). Commonly they work to both build capacity within existing institutions and to facilitate the development of new civil society institutions, which in turn can promote public-private coordination (e.g. Case Study 1, Appendix E.1). Through these activities, a CTF should over time substantially lower the transaction costs of PES operation (RedLAC 2010b). For instance, CTFs have potential to help to address the widespread limiting factor of weak enforcement in developing-world fisheries PES. They can help to build capacity for monitoring and enforcement within local organisations and government agencies (e.g. Case Studies 8 and 11, Appendix E.1), or facilitate the development of community-based institutions and hybrid co-management structures (Case Study 11, Appendix E.1), which should improve management and reduce transaction costs by increasing the congruence between local and national interests (Sen & Nielsen 1996; Carlsson & Berkes 2005; Berkes 2009, 2012; Carbonetti et al. 2014). These hybrid structures are commonly found in fisheries, but often lead to a shift in responsibility without this being reflected in updated regulations. When they lack empowerment capacity, hybrid institutions can actually serve to reinforce local elite powers or strengthen state control (Jentoft 2000, 2005; Nielsen et al. 2004). With independent CTF facilitation and capacity-building, this issue could be ameliorated.

Although CTFs rarely make grants directly to individuals, CTF grantees often reallocate funds to local communities and individual households (e.g. Case Study 2, Appendix E.1). As a result of their capacity-building and facilitation activities, CTFs were noted by some KIs (AF1; MAR1; CF3) for their potential to serve as efficient and effective mechanisms for channelling payments down to local communities through PES (see Case Studies 2 and 10, Appendix E.1). In a community-based PES, a CTF could facilitate open and equitable negotiations on how to invest or spend the payment. By shielding the CTF from individual or political agendas, a diverse and participatory governance structure should enable the CTF to resist the elite capture sometimes typical of state and market mechanisms and increase fairness in benefit distribution (CF1; CF5; Spergel & Wells 2009). Although developing-country fishing communities often lack prior organisational experience, a CTF could facilitate their organisation into smaller sub-groups more suited to self-management (AF1). For example, the Mesoamerican Reef (MAR) Fund runs a Community Fisheries Programme in support of the active participation of self-organised groups of fishers in PA management (see Case Study 6, Appendix E.1). In one of its focal countries, Guatemala, fishers organised themselves into co-management associations, in response to the lack of financial resources within Government for control and surveillance in no-take zones, and now receive support from MAR Fund for their own control and surveillance activities. In another – Honduras – fishing communities have created an association and co-management agreement and identified training and capacity-building needs for which they are given assistance from MAR Fund. There is a limit to the extent to which a CTF could address weaknesses in the characteristics of users within these groups, but it could potentially provide support for training, assistance in the identification of appropriate leadership, and facilitation of improved social cohesion. The ability of a CTF to influence the capacity of providers to change their relationship with a resource depends upon the level of this capacity in the first place; it might enable them to change their behaviour through providing compensation, alternative livelihood assistance, or microcredit, but if they are too poor and dependent upon the resource, this assistance may have limited impact. If providers lack secure tenure or use rights, a CTF could play a role in lobbying governments for these rights and facilitating their clarification, especially if influential politicians or corporate leaders are represented on the board (Spergel & Taieb 2008). The MAR Fund, for instance, has supported the implementation of fishing rights in one PA in Brazil. Furthermore, the CTF model allows continued support and assistance beyond the usual short project funding cycles (CF1; CF4), which should enable greater institutional sustainability than if an NGO was acting in this intermediary capacity (e.g. Case Study 2, Appendix E.1).

Best practice standards also direct that a CTF should coordinate reporting, monitoring and evaluation at four levels – grantee, CTF, donor and government – in the countries where the CTF is registered or operates (Spergel & Mikitin 2013). It should monitor and evaluate the impacts of its grants in relation to its mission and strategic plan, and in relation to national or site-level management plans and biodiversity indicators, targets and strategies. Through this requirement, grantees must develop goals and indicators for biodiversity conservation in their proposals, collect relevant baseline data and submit this multiple times during implementation and after grant completion, while maintaining strong financial accountability through internal and external audits (Spergel & Taieb 2008). The CTF must therefore provide clear financial reporting and social and ecological monitoring protocols to the grantees; make sure the protocols are transparent for both buyers and providers; and if the grantees lack technical or financial capacity for these activities, the CTF should help to build capacity (Case Studies 6 and 12, Appendix E.1). For a PES, a CTF should thus provide the means to bring all of the relevant stakeholders together, not only for the design of the PES contract, but for the design of a rigorous monitoring and evaluation programme from the very beginning (CF4). Its multi-stakeholder nature should also provide more opportunity for collaboration and information sharing than a state- or user-run PES, and more opportunity to harness international expertise (Case Study 10, Appendix E.1). Ultimately, though, establishing additionality and tracing this back to management will depend on whether *ex-ante* counterfactual scenarios are incorporated into evaluation – a process that is still rare in PES (Pagiola & Rios 2008, 2013). FONAG's PES, for instance, has suffered from a lack of science linking payment distribution to ES provision (CF2; Case Study 10, Appendix E.1).

It should also be noted that although CTFs tend to do a good job of monitoring institutional performance, monitoring and evaluation of actual conservation impact is still a challenge (CF2; RedLAC 2008; Spergel & Taieb 2008), and a common criticism from KIs (FMCN1; FMCN2; CF1; PACT1) was a lack of due diligence and rigorous assessment of grant impacts (e.g. Case Studies 8 and 12, Appendix E.1). For large CTFs, particularly those with objectives broader than PA management, it can be difficult to aggregate results from individual grants, and mature CTFs suffer from not having designed projects with monitoring and evaluation in mind. The newer

CTFs (e.g. Case Study 11, Appendix E.1) are, however, taking a more systematic approach (CF2; RedLAC 2012).

A CTF cannot address many of the key challenges in PES that arise from the resource system itself (feasibility of improving the resource, manageability of the resource, or potential for profitability at a sustainable rate). KIs (CF3; CF6) also noted that CTF engagement in PES will not always be appropriate, but is dependent on the CTF's strategic plan and capabilities. Others (FMCN1; CF1; CF2) emphasised that the creation of a new CTF is time-consuming, tedious and expensive, and therefore not always the best option, especially if a limited amount of capital is available and if there are existing legally established mechanisms or institutional frameworks available for PES. The PIPA Trust, for instance, took a long time to establish itself in the absence of a legal framework for marine conservation and CTFs, and when Arannayk was established, eight years passed before it was able to start making grants (see Case Studies 2 and 3, Appendix E.1). Given that some larger CTFs administer more than one fund (e.g. Case Study 1, Appendix E.1) it may in some circumstances be preferable to use a pre-existing CTF to administer a PES, rather than creating a new one.

7.3.3 Sustainability of the hilsa fisher compensation scheme in Bangladesh

Here I examine the extent to which the enabling conditions for sustainability are present in the current compensation scheme for hilsa fishers in Bangladesh (Fig. 7.2; Table 7.6). The ecological status of the Bangladesh hilsa fishery is unclear, but there is evidence to suggest that stocks have declined and that they will continue to do so with projected climate change, under current management (Chapter 3; Fernandes et al. 2015). As a valuable commercial fishery, there is potential for profitable exploitation at a sustainable rate. However, in Bangladesh there is a lack of fishery-independent stock assessments and an absence of adequate and reliable baseline data against which to evaluate management interventions (Chapters 3 & 5; DoF 2002). This is a result of limited financial and technical capacity for monitoring within government agencies, combined with weak coordination and information sharing amongst stakeholders (Chapter 5;

Islam et al. 2016), and makes it very difficult to attribute any apparent trends in the hilsa fishery to management, or even to assess the scope for impact of a PES (Chapter 5). Although the hilsa sanctuaries (see Chapter 3) might be described as 'small', their manageability is low since the hilsa within them travel far beyond their boundaries. As a migratory species the hilsa stock spans a vast area of marine and freshwater within Bangladesh, India and Myanmar (Salini et al. 2004), complicating the attribution of trends.

The institutional arrangement is currently one of top-down governance, both in terms of design and implementation, but a common theme in KIIs was the importance of creating institutions that facilitate self-management by the fishers, with support from the Government (A2; D3; D4; D5; LN2; IN4). The hilsa fishery comprises a large and dispersed group of fishers – between 300,000 and 500,000 (Haldar 2004; Rahman et al. 2014a; Islam et al. 2016) – whose fishing grounds overlap. Despite the top-down governance, it is largely open access (licenses are required only in the marine sector and effort restrictions only affect trawlers) with a long history of customary rights in some areas (A. Bladon 2015, personal observation; Dastidar 2009).

It is feasible that the fishers could be split into smaller subgroups for self-management. Indeed, some of the flaws in the current compensation scheme may stem from the fact that it is being centrally managed across such a large area. What may be more sustainable is a scenario where multiple locally-managed projects are in place, which could then inform regional or national management (Greiber 2009). Village leaders tend to make community decisions (Islam et al. 2016), and some local associations also exist in hilsa communities (F1-3; Rahman et al. 2014a). However, these associations were described as 'non-functional' by local NGO KIs (LN3; LN5); they said the leaders lack trust from the communities and, with no technical or financial support, they lack the capacity or legal status to perform any useful organisational or governance role. A community-based fisheries management project has had success in some inland water bodies in Bangladesh, regarding the development of local institutions and management capacity within them and the empowerment of poor fishing households, but outcomes have been limited by complex power relations, politicisation of NGOs, limited resources, capacity of the Department of Fisheries (DoF) at the field level, and limited motivation of local administrations (Béné & Neiland 2006; Rab 2009). Rab (2009) noted that the community-based management approach was much more challenging in flowing rivers than closed water bodies, in terms of distinct boundaries, clear tenure and proximity of resource users to the resource. Coastal areas would present similar challenges; managing a community fishing from a distinct lake or pond is very different from managing a coastal community or a landing site where the catch could have come from much further away. A KI scientist (A2) said that these challenges are 'not insurmountable', and that progress in community-based management of rivers and open water has been limited so far by funding opportunities rather than practical potential – an opinion backed up by a local NGO KI (LN4). Community-based management of hilsa sanctuary areas has been proposed in the hilsa fishery management plan, where management committees would be composed of 10-15 members from each level of local Government, public and fishers (DoF 2002) – although this does not appear to have been implemented yet.

Inland fisheries management experience suggests that the development of appropriate local leadership for hilsa management would also be challenging. There are numerous local and international NGOs working with hilsa communities that could provide leadership and facilitate the engagement of marginalised populations (Huber-Stearns et al. 2013; Escobar et al. 2013), but the complexity of power relations means that internal factions can develop when NGOs try to promote transparent and accountable leadership (Béné & Neiland 2006). Similarly, there is some heterogeneity within fishing communities that could limit the opportunity for collective action. For instance, KI A2 noted major conflict in the Bay of Bengal between artisanal hilsa fishers and the industrial fleet. Moreover, although poor fishers form the majority of residents, there are often traditional fishers[31], full-time but non-traditional fishers[32], seasonal or opportunistic fishers, relatively wealthy fishers and middlemen who own boats and nets and lend money, all living in the same communities and each with different interests (LN1; Rab 2009). Local elite boat owners and other middlemen are widely reported to exploit fishers and drive unsustainable and illegal fishing practices, and therefore fishers do not trust them (D4; LN2; LN3; Chapters 3 and 5). However, some NGO KIs (IN6; LN4) saw potential for these 'big shots' to harness the power and influence they have over fishers to encourage sustainable behaviour. As one fishers' association member said, middlemen are 'brothers of fishers' and fishers are more likely to listen to them than to Government (F2). The development of flexible and inclusive local institutions may help different stakeholders to negotiate their interests and allow local elites to work with fishers in a more positive way.

There is also a complete lack of trust between fishers and Government in Bangladesh, which is a significant institutional barrier to the sustainability of the current Government-financed compensation scheme (M1; M3; M4; M5; LN2; LN3; LN5; LN6; F2; F3). Compensation is distributed via local Government in a system that lacks transparency and accountability, and officials are notorious for corruption (Chapter 6; Haldar & Ali 2014). This lack of trust also extends to other enforcement authorities, which are reported to harass fishers and take bribes, reducing the potential for sanctions to provide an effective deterrent (L1; L2; D3; F2; F3; Chapter 6). A more sustainable PES would require substantial intermediary facilitation to build trust if the Government is to remain a major buyer, but hopefully this trust would improve if communities were given more autonomy.

Hilsa fishers tend to be very poor and highly dependent on fishing for income and livelihoods (Chapter 6), which means that although there may be strong community interest in conservation, there may be low capacity to change the relationship between hilsa fishers and hilsa – a hypothesis validated by the widespread lack of compliance with rules (A4; Chapter 5).

[31] Fishers from birth – usually Hindu.
[32] Usually Muslim.

The current compensation scheme was no doubt initiated through recognition of this dependence (Chapter 5), but compensation distribution is currently neither fair nor transparent, and is heavily politicised (Chapter 6; Haldar & Ali 2014b). Only 36.2 per cent of fishers interviewed in a recent survey said they think that the distribution of compensation is fair, and they perceive a high level of elite capture (Chapter 6). With community participation in scheme design and implementation, it is likely that perceptions of fairness would become more positive (Marshall 2005; le Coq et al. 2015).

The form in which the incentives are received at the household or community level is beyond the scope of this chapter, although there is some evidence to suggest that the current form is not the most appropriate (Dewhurst-Richman et al. 2016; Chapter 5). Some KIs supported cash or alternative livelihood support as more appropriate payments (LN3; LN4; LN5; IN6), one pointing out that if it was monetary, the use of MobiCash[33] would be the most transparent method of distribution (IN6). Another proposed that a low-interest loan conditional on compliance would be most suitable, but only if repayments were frozen during fishing ban periods (LN2).

Given that artisanal hilsa fishers currently have no formal tenure or use rights, for a sustainable PES they would require organisation into formal Community-Based Organisations (CBOs), which may also entail the development of legal contracts for community-level fishing rights. Multiple KIs (A2; D4; IN4; IN5; LN3) voiced opinions that hilsa management would be more effective if fishers were to 'own' sections of river (although this would be much more complicated in marine waters). The river fisheries have in the past been through various systems of licensing, but access was repeatedly given to the highest bidders rather than genuinely dependent fishers, even when the DoF collaborated with NGOs (Rab 2009; Rahman et al. 2015), and so future efforts must be more equitable. Care should also be taken to consider pre-existing customary rights (Cinner & Aswani 2007).

[33] A secure cashless mobile financial platform that anybody with a mobile phone can use, including unbanked citizens (http://www.mobicashonline.com/)

Overall, the hilsa fishing rules are simple and communicated through awareness-raising activities (Chapter 5), but it is unclear exactly which behaviours are the ones which are compensated for and which are not. Compensation is provided during the peak period of *jatka* presence – beyond the sanctuary fishing bans but not so long as to cover the entire *jatka* ban – and so it is unclear whether fishers are being compensated for abiding by the sanctuary fishing bans or for abiding more generally by the *jatka* ban (Chapter 5). In general, DoF activities are not transparent – one local KI (A2) criticised it for a fast turnover of staff that results in 'forgotten pockets of money' – and this lack of transparency is apparent in the administration of the compensation scheme (Haldar & Ali 2014a). Another major institutional weakness is the poor enforcement of rules; anecdotal reports from a range of sectors, including the DoF itself, indicate a lack of monitoring, particularly in inland areas, and sanctioning mechanisms – which are quite rarely implemented – often involve delays in application and are ineffective as a deterrent (Chapter 5). The sanctions are also unfair; as one KI (LN4) put it, 'it is the very poor fishers who are caught, but their fishing is financed by the rich who are never caught'. Furthermore, although the compensation scheme is often referred to as a PES (e.g. Islam et al. 2016), the lack of enforcement and therefore of conditionality limits potential for the payment to act as an incentive.

Under Government control and financing, the sustainability of the compensation scheme is dependent on the political climate and on financial resources available to the Government. The transaction costs of the compensation scheme are currently higher than they need to be, largely due to the complex way in which compensation is distributed (Haldar & Ali 2014b). The process of restructuring the scheme for improved sustainability (e.g. through establishing CBOs and building capacity for self-management, clarifying rights, improving fairness in benefit distribution and collecting information on hilsa) could substantially raise transaction costs, and it is unlikely that the Government alone would realistically cover these costs. As a KI DoF official admitted (D1), it does not expect to be able to maintain its budget allocations to the compensation scheme beyond 2025. External intermediary involvement would therefore be vital to cover transaction costs.

The pre-existence of the compensation scheme could provide some institutional basis for the development of a PES, but it could also cause problems (Corbera et al. 2009; Huber-Stearns et al. 2015); it might have planted the seed and helped develop the political will for PES within the Bangladesh Government, but distrust of the Government among fishers may also taint future efforts. There is no articulation of the scheme with hilsa markets, and although the prospect of an eco-label for hilsa came up as a real possibility in one KII (IN6); others, including a member of the Government's Export Promotion Bureau[34] (A2; C2; IN2; A5; LN1), said that a consumer price premium has limited scope, given the current lack of traceability in the supply chain, the ban on hilsa exports from Bangladesh, and the relatively niche, although sizeable, export markets (India and areas of Europe and the Middle East where there are Bengali diaspora; see Chapter 3). However, the market in Bangladesh alone is commercial and large, and if some resources were directed into supply chain traceability, this could be an opportunity for improved sustainability in the future.

[34] Government agency within the Ministry of Commerce and responsible for Bangladesh's export industry.

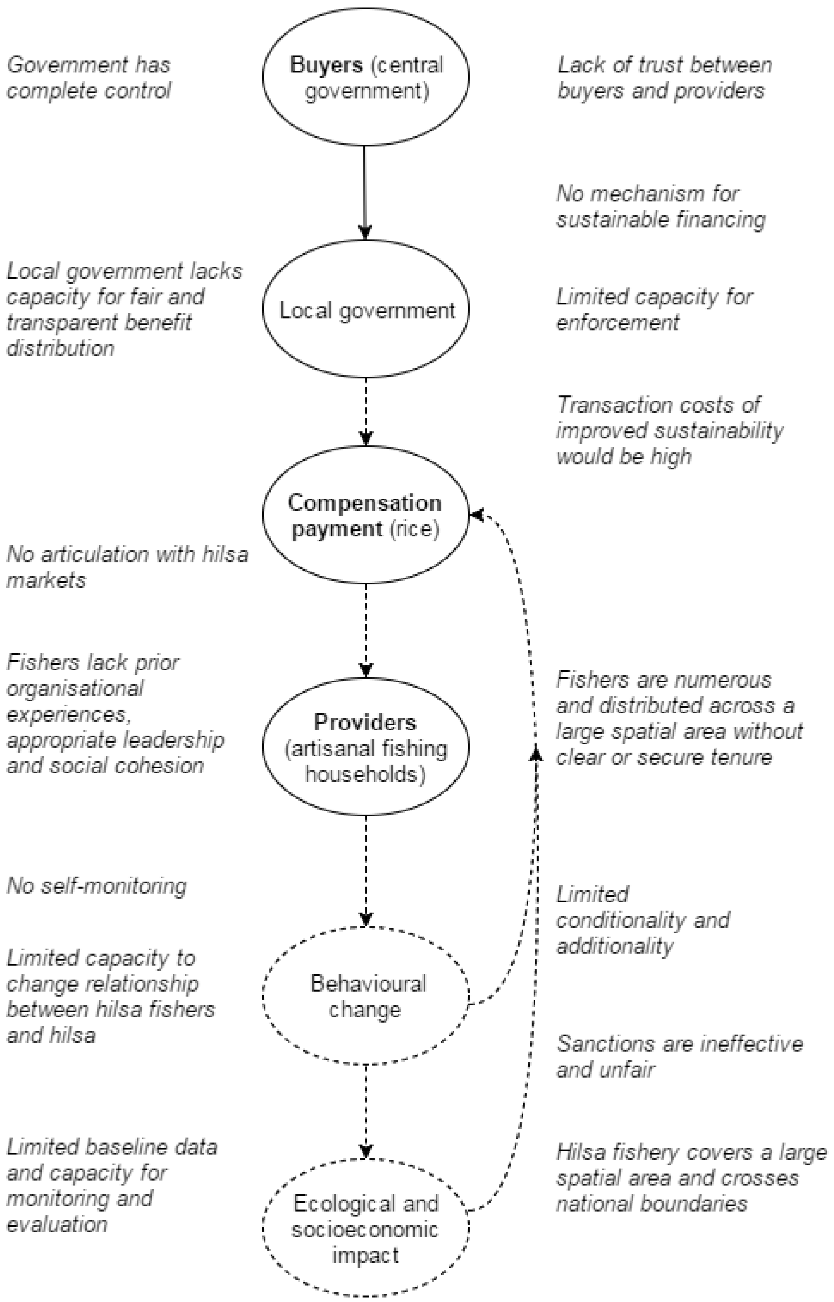

Figure 7.2: Conceptual diagram showing the institutional framework of the current compensation scheme for hilsa fishers in Bangladesh. Italics and dotted lines represent the weaknesses currently limiting the outcomes that are desired from a sustainable PES.

7.3.4 Could a CTF catalyse sustainable PES in the hilsa fishery?

Here I explore whether a CTF could provide a framework for a more sustainable hilsa conservation payment scheme in Bangladesh (Table 7.6). It should be noted, however, that the extent to which it could do so depends largely on which financial and governance structures are feasible for a CTF within the institutional context of Bangladesh, and how well the CTF is implemented (Fig. 7.3). Under best practice, the involvement of a CTF could address the current lack of a mechanism for financial sustainability. Through a diversified funding strategy and low-risk investment, a CTF could guarantee stable revenues for the PES and associated hilsa management in the long term, allowing more funding to be allocated to capacity building and monitoring. The creation of a new CTF would first require seed capital from the Government and/or donors. Initial capital required has been roughly estimated at USD 29 million (Majumder et al. 2015a), but an experienced KI stated that the minimum threshold of capital required to establish a CTF generally lies between USD 5-10 million, and that it is rare that a national fund with such a narrow mandate would start with more than USD 10 million (FMCN1). Arannayk, a CTF established in 2003 (Case Study 2, Appendix E), for example, started with only USD 8.5 million. Given previously cited CTF KI comments, it might be preferable for an existing CTF that is capable of managing multiple funds under a single legal and institutional structure to administer the PES, rather than creating a new one. If there is limited capital available, this would be more cost-effective, and should result in grants being disbursed for hilsa conservation earlier (FMCN1; CF1; CF2). The only truly independent CTF in Bangladesh is Arannayk, which primarily focuses on terrestrial conservation. A KI representative (AF1) did propose that it would be in a position to take on the role, but this was a hypothetical discussion and there may of course be disadvantages to administration by a CTF that does not have hilsa conservation as its primary goal.

Ideally, the CTF would manage an endowment together with a revolving fund that could be financed through hilsa supply chain beneficiary fees (e.g. boat owners, middlemen, processors, local distributers, exporters, hotels and restaurants). It would not currently be feasible for external markets to become buyers due to the lack of traceability and, since PPPs are still a new concept in Bangladesh (T1; T3; T4), voluntary investment from supply chain companies was also deemed unlikely at this stage (IN6). This might, however, be a possibility for the future. When supply chain KIs were asked whether they would be willing to contribute, the boat owner

(F1) said he felt no responsibility to do so, but middlemen (M1; M2; M3; M4; M5) were more supportive of the prospect (in general they tend to be relatively more wealthy than boat owners) – some on the condition that corruption within the Government is reduced and others on the condition that the funds would not be handled at all by the Government.

NGOs have proposed that export taxes could be utilised (Majumder et al. 2015a). The Government has in the past reported revenues of up to USD 45 million per year from hilsa exports (Chapter 3), and if 5-10 per cent[35] of this were earmarked it could theoretically provide a sustainable funding stream for hilsa conservation. However, local KIs (C2; LN1; LN3; LN4; T5; D1) doubted the practicality of this approach since there is currently a politically-motivated export ban in place that is unlikely to be lifted (see Chapter 3). A representative of the Export Promotion Bureau (C2) said hilsa export earnings are currently 'insignificant' and that hilsa would simply be smuggled into India to avoid taxation, as it has been during the ban. There might be scope to tax the industrial trawl fleet through license fees or landings revenues (LN1), although its catch might not be large enough to generate a significant amount (LN1), and this 'would not go down well with the political businessmen' (A2).

Funds could also be solicited from existing climate change Trust Funds in the country, since hilsa are likely to be heavily impacted by climate change (Chapter 3) and therefore fall under their remit. Other potential funding mechanisms include debt service liability, where the Government would repay principal and interest for a loan from development partners (Majumder et al. 2015a), and impact investment – typically debt or equity investments that target social and, increasingly, environmental issues in developing countries (Bos et al. 2015). Impact investment in sustainable fisheries is a new strategy pioneered by a group of philanthropic organisations, NGOs and investment firms, some of which have developed hypothetical investment models that incorporate the use of financial rewards to fishers for

[35] A 'small amount', as suggested by (Majumder et al. 2015a).

sustainable practices (Manta Consulting 2012; Encourage Capital 2015)[36]. Impact investment is an approach that could be useful at a later stage when initial plans have already been funded by donors and philanthropists and so risks are lower (CF7).

In terms of governance structure, the independence of the CTF should require control of the PES to be decentralised from Government, and this would have numerous advantages. First and foremost, decentralisation should shield the PES from political whims, ensuring continuity under changing Government priorities. It should also create an environment that empowers fishing communities to self-organise and have some participation and autonomy in setting and enforcing rules, without being overruled or undermined by Government. This could help to avoid the current politicisation in social targeting and elite capture of benefits, thereby improving perceptions of fairness. The extent to which a CTF could address the issue of fairness may be limited in Bangladesh, however, where corruption is high and present in every sector (UNCAC 2011; Transparency International 2012).

A CTF may also help to build trust between buyers and providers. As observed in conversations with hilsa supply chain and local NGO KIs (M1; M3; M4; M5; LN2; LN3; LN5; LN6; F2; F3), fishing communities do not trust the Government – a problem which is related to perceptions of fairness in benefit distribution and approaches to enforcement. By shifting power from the hands of the Government, a CTF should be able to build trust with providers, no matter who the buyers are. Similarly, decentralisation could potentially allow the CTF to engage with buyers who otherwise would not be interested. Donors in particular may be attracted by the increased transparency.

In a decentralised PES, grants would ideally be channelled directly to providers, but given that hilsa fishing communities are largely unorganised and existing organisations generally lack the

[36] These models from Encourage Capital (2015) range from small-scale fisheries investment (with a focus on supply chain infrastructure and operations, creating small- to medium-sized enterprises and implementing management improvements to support their sourcing needs) to industrial-scale fisheries (similar, but bundling fisheries improvement investments with investments in fishing assets and seafood companies, to generate cash flow tightly tied to stock recovery), to national scale public-private investment (with a focus on fishery-wide data collection and port infrastructure).

capacity to act as grantees, potential initial grantee candidates are more likely to be local NGOs or local Government units. According to a local Government KI (L4), NGO grantees would be preferable to local Government units, which lack the capacity to write good proposals and to keep track of fishing households for compensation distribution, whereas NGOs running disaster relief and social safety net programmes have transferable experience in channelling funds to the community level (T2; Pham et al. 2010). Through these local intermediaries, a CTF could facilitate the identification of hilsa fishers and their organisation into smaller sub-groups (CBOs) more suited to self-management – a process undertaken by Arannayk in remote terrestrial areas of Bangladesh (see Case Study 2, Appendix E.1) – and lower the transaction costs of doing so. Although the hilsa fishery is at a much larger scale than the communities organised by Arannayk, lessons could be transferred to small-scale pilots.

Multiple KIs (T1; T5; AF1; A2) emphasised that CBOs require a legal basis to be fully functional and so should be registered under the Societies Registration Act, 1860. If monetary, collective payments could be managed as revolving funds, as is the case in the MAR Fund's Community Fisheries Program (Case Study 6, Appendix E.1) and either channelled down to individual households in an appropriate form or used collectively by the community. Other Bangladesh Trust Fund KIs (AF1; T1; T5) directed that this should be done in a participatory way, and after a certain length of time villagers could be aggregated at district and sub-district level, so that when intermediaries withdraw their assistance the village-level institution can sustain payments and management. Some KIs (A2; IN5) pointed out that lessons could be taken from wetland management in Bangladesh, where the Government allocated endowment funds to multi-stakeholder groups who were helped by NGOs to establish wetland sanctuaries. CBOs were then formed around these sanctuaries and livelihood support was provided to resource users for not catching breeding fish.

Under a CTF model, CBOs should be more sustainable than if they had been formed by NGOs or Government alone. Trust Fund KIs (T1; T5) pointed out that NGOs are, contrary to their name, often profit-earning in Bangladesh, and tend to completely withdraw both funds and support at the end of project cycles, while Government funds are usually subject to long bureaucratic delays. For example, in 2003 a local NGO received funds from the DoF to pilot an alternative livelihoods project involving the formation of community groups for fishers in Chandpur district. Following a long delay by the DoF in transferring the remaining funds, said KI LN3, the groups have not yet been registered as CBOs with a long-term management plan and the project has stalled because there is no endowment fund for phasing out support. Yet, although they are relatively few, some bureaucratic delays have still been experienced by Arannayk (see Case Study 2, Appendix E.1), indicating that in Bangladesh no institution is completely free from such bureaucracy.

The hilsa fishery management plan proposed over a decade ago that fishing rights should be granted to fishers participating in management of the hilsa sanctuaries (DoF 2002), demonstrating some political will for contracts of community-level fishing rights. A multi-stakeholder CTF could potentially lobby for and empower CBOs to negotiate for this tenure, speeding up the process. Short-term property rights (three years) are endowed by the Government to CBOs in wetland fisheries, but the process of developing the collective action to form these CBOs is dependent on political connections and financial capacity, rather than on resource dependence (Rahman et al. 2015; IN5). Although in theory a CTF should facilitate more permanence and fairness in the process, there is little to suggest that the same thing wouldn't happen under a CTF.

Although the CTF approach cannot necessarily address the issues surrounding extreme poverty or high dependence on hilsa, by enabling improvements in the current compensation scheme, and perhaps generating more resources for tandem livelihood or social assistance, it could potentially contribute to a reduction in fishing dependence and poverty, thereby raising the capacity for change in resource use.

A CTF could help to address the current lack of technical and financial capacity for monitoring hilsa in Bangladesh, reducing the transaction costs of collecting information. As Arannayk has done in forest areas (Case Study 2, Appendix E.1), the CTF would assist CBOs in defining the scope of monitoring and evaluation, the level of engagement of local communities and the participatory identification of appropriate social and ecological indicators. Training and logistical support could be provided for the collection of baseline data, and the requirement for CTF grantees to have a strong system of monitoring, reporting, and, therefore, accountability should contribute to transparency for buyers and providers, and thus trust (McClanahan et al. 2013). For instance, MAR Fund provides its community fisheries with financial training (Case Study 6, Appendix E.1), and Arannayk not only has an annual external audit, it reviews activities every three months, lowering the risk of misappropriation of funds (Case Study 2, Appendix E.1).

Depending on the size of its endowment, a CTF could provide more financial and logistical assistance than the DoF is able to for the communication and enforcement of rules, allowing conditionality to be established in a PES. The community ownership could also play a role in the development of self-monitoring as a social norm, which in turn should reduce reliance on corrupt and ineffective Government enforcement. If CBOs could enforce their own sanctions (for instance, removal of fishing rights) then the power of Government officials to abuse the sanctioning system may be reduced. A CTF would not be in a position to directly address the limitations to PES in terms of provider group characteristics, but it could help to bring local intermediaries together to encourage social cohesion and identify local leadership. For example, AF2 pointed out that since NGOs in Bangladesh are heavily involved in microfinance, many would not welcome the idea of a community managing their own funds from a PES. Similarly, through PES, middlemen could lose the financial power that they have over fishers. But a CTF could mediate between these divergent interest groups, and by building institutions for collective action, trust and social cohesion should in turn improve. This could in turn allow elites and middlemen to take a more positive leadership role for collective action.

However, a CTF could not address some of the key challenges of community-based management of the hilsa fishery, in terms of overlapping fishing grounds, proximity of fishers to fishing grounds, and unclear boundaries. Even with the advantages that a CTF provides, self-monitoring is less likely to work in coastal areas than inland. Furthermore, a national CTF cannot directly address the fact that hilsa are not a small-scale resource, and that a PES could be undermined by transboundary movements of both hilsa and fishers. Since hilsa are a transboundary resource, a regional CTF (between Bangladesh, India and Myanmar) would be most appropriate. Yet, with no history of regional cooperation, KIs saw this as unnecessary, giving answers like 'What is important is that we protect our territory' (R1) and 'India are the big brothers so [they are] difficult to manage' (LN5). A regional arrangement might nevertheless be possible in the future and large-scale regional CTFs like MAR Fund have in fact benefited from the experience and coordination of pre-existing national funds (Case Study 5 and 6, Appendix E.1).

An ex-Government KI (LN6) also pointed out that the concept of a Trust Fund is fairly new in Bangladesh, and although various kinds exist, with the exception of Arannayk they are all public. Since PPPs are also a new idea, a Trust Fund representative pointed out that the Government is unlikely to channel funds into a CTF in which it does not hold the majority position (T3). The absence of a PPP tradition can make it difficult for governments to accept the principle of mixed management where they do not hold the major power position (see Case Study 7, Appendix E.1) and difficulties are often encountered in these circumstances when CTFs intend to mobilise resources through public revenues such as taxes (see Case Study 11, Appendix E.1). As a result there are CTFs that stray from best practice, some of which have achieved their goals nonetheless (e.g. Case Study 8, Appendix E.1), but without government independence a CTF is unlikely to provide a sustainable institutional framework for PES.

The overwhelmingly common Bangladeshi KI view on CTF governance structure (from a range of organisations at both national and local levels; T1; T2; T3; T4; T5; LN1; LN3; LN5; LN6; D3; D4; C1; IN2; IN3; A3; A4) was that Government ownership of and support is vital – 'anything with a long-term perspective must be embedded and tied up in a Government mechanism' (T3). They did mostly also express that there should, nonetheless, be representation from the fishing communities in these circumstances; without some ownership themselves fishers will see any payment as relief from the Government, just as the compensation is seen now. However, this view indicates that a Government-dominated board might currently be the only viable approach, with a view to eventually becoming increasingly independent.

At the time of writing, a CTF for hilsa conservation is going through the ratification process, which would be housed within the Ministry of Fisheries and Livestock and would have a Government-dominated board. It is proposed to provide a small fund to each *upazila*[37], which would then be managed by local Government committees for use by community-based hilsa management organisations (Majumder et al. 2015a). Although this might be the most feasible institutional structure for a hilsa CTF currently, there is a risk that it would simply try to fund business-as-usual, without an improvement in sustainability of the compensation scheme (Fig. 7.3b).

[37] Subdistrict

Table 7.6: Enabling conditions for sustainability of a PES in developing-world fisheries, with ticks and crosses showing whether these are fulfilled in the case of the Bangladesh hilsa fishery, whether improvements are required, and whether a CTF could support these improvements, given the current institutional context of the hilsa.

Enabling conditions	Fulfilled in Case Study	Improvements required	Can a CTF support these improvements
Resource system			
Feasibility of improving resource	✓	✗	✗
Indicators of resource conditions	✗	✓	✓
Traceability of resource improvement	✗	✓	✓
Manageability of the system	✗	✓	✗
Potential for profitable exploitation at a sustainable rate	✓	✗	✗
Group characteristics			
Small resource user group	✗	✓	✓
Prior organisational experience within resource user group	✓	✓	✓ ✓
Appropriate leadership within resource user group	✗	✓	✓
Social cohesion within resource user group	✗	✓	✓
Trust between stakeholders, particularly between buyers and providers	✗	✓	✓
Relationship between resource and group characteristics			
Capacity to change relationship	✓	✓	✓
Fair and transparent benefit distribution	✗	✓	✓
Resource tenure clarity and security	✗	✓	✓
Institutional arrangement			
Rules are simple and transparent	✓	✓	✓
Effective enforcement	✗	✓	✓
Self-monitoring	✗	✓	✓
Swiftly applied, graduated, fair sanctions	✗	✓	✓
Low transaction costs	✗	✓	✓
Mechanism for sustainable financing	✗	✓	✓
External factors			
Central government provides a supportive environment without undermining local autonomy	✗	✓	✓
Appropriate levels of articulation with external markets	✗	✓	✓

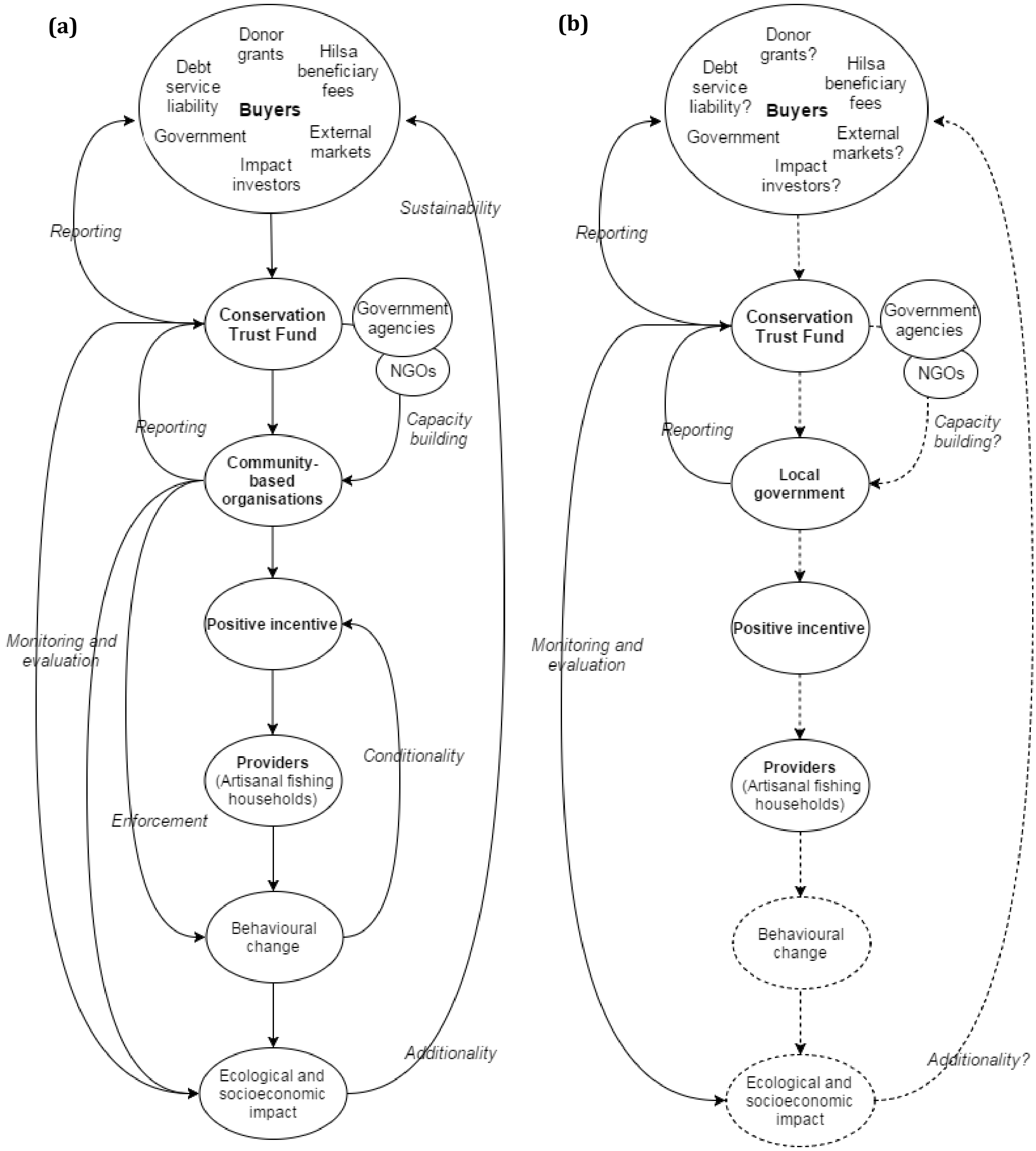

Figure 7.3: Conceptual diagrams showing **(a)** how a best practice CTF could enable sustainable PES in Bangladesh; and **(b)** how this framework would differ without complete best practice, based on the analysis of the current situation for hilsa in Bangladesh (dotted lines and question marks represent areas of uncertainty introduced by probable gaps in best practice, given the current institutional context).

7.4 Discussion

The sustainability of conservation payments is heavily dependent on their institutional structure and wider institutional context, and calls have been made for greater attention to be placed on how this structure and context influence the motivation and capacity of providers to manage their resources (Corbera et al. 2009; Vatn 2010; Primmer et al. 2013; Hayes et al. 2015). As seen in the Bangladesh hilsa fishery, social-ecological conditions in coastal developing-world fisheries can limit the development of sustainable PES institutions (Bladon et al. 2014b). In particular, they can lead to a lack of monitoring and enforcement, and therefore of compliance, which constrains the key elements of conditionality and additionality in PES (Sommerville et al. 2011; Pham et al. 2015).

To our knowledge, this chapter is the first piece of research to examine the potential role of CTFs in catalysing and supporting sustainable payment schemes, and specifically in advancing the conservation and sustainable management of developing-world fisheries. Some members of RedLAC (the Latin American and Caribbean Network of CTFs) are already active participants in, or developing the capacity to participate in, PES; but although the valuable role of PES in generating sustainable financing for CTFs is clear, the whole range of potential benefits have not been recognised (RedLAC 2010a). This research demonstrates that although a CTF model cannot provide all of the enabling conditions for sustainable PES, under best practice it can act as much more than a mechanism for disbursing payments, by helping to catalyse the majority of these enabling conditions. It could thereby provide a way to implement sustainable payments in a developing-world fisheries context, despite the inherent complexities, uncertainties, and limitations which these types of resource systems pose. The CTF model may be particularly advantageous where existing implementing institutions lack capacity, in politically unstable situations, and when current payment structures lack financial sustainability (Goldman-Benner et al. 2012). Most importantly, the CTF model could allow additionality and conditionality to be established in PES which lack resources for conventional monitoring and enforcement.

However, this potential all depends on actual implementation; if best practice is not followed then a CTF will be no more capable than any other institution of addressing limitations to sustainability in PES. Although there are numerous examples of effective best practice CTFs – some of which are administering PES schemes – there are also examples of CTFs with poor governance, lack of financial planning, weak systems of monitoring and evaluation, and capitalisation issues (Bladon et al. 2014a).

In Bangladesh, institutional and financial conditions currently limit the potential for development of the hilsa fisher compensation scheme into a more sustainable PES (Table 7.7). As is the case in many developing countries (Pham et al. 2015) the existing institutions for hilsa management lack the capacity that PES requires. The involvement of a legally independent multi-stakeholder CTF would, in theory, require decoupling of the scheme from the control of central Government and, under best practice, this should facilitate capacity building within existing institutions and, where appropriate, the development of new or evolved civil society institutions (Fig. 7.3a). This could, in turn, empower communities and encourage collective action for decentralisation of management, building trust, lowering transaction costs, and potentially allowing conditionality and additionality to be established.

Yet, in practice, truly independent governance of a hilsa CTF may be an unrealistic expectation. Given the institutional context and the Bangladesh Government's interest in the hilsa fishery, initially at least, majority Government ownership might be required for the CTF to be feasible. The current CTF proposal (Majumder et al. 2015b) could provide an institutional framework that simply maintains the status quo of the compensation scheme, with limited improvements in sustainability (Fig. 7.3b), although it may nevertheless have potential for impact in the future (Table 7.7). If best practice governance standards cannot be followed initially, mechanisms should at least be in place to ensure that the governance structure can evolve with time, as seen in BTFEC (Case Study 8, Appendix E.1).

Table 7.7: Summary of the major challenges for a Conservation Trust Fund (CTF) in providing a sustainable framework for hilsa conservation payments in Bangladesh, and recommendations for how these challenges could be addressed. CBO = Community-Based Organisation.

Major Challenges	**How these challenges could be addressed**
• Independent governance	• Ensure that there is a mechanism in place for the board to become increasingly independent from the Bangladesh Government
• Collective action for the formation of CBOs and endowment of fishing rights could be heavily influenced by financial and social capital	• Initially, grants should be focused on helping communities to form CBOs and reducing elite capture and other inequities in the process • Ensure governance of the CTF is not Government-dominated • Take lessons from other Trust Funds in Bangladesh with experience in channelling payments to local communities
• Capitalisation of the CTF	• Use an existing CTF that is capable of administering multiple funds • Ensure best practice standards are followed in the creation of a CTF to maximise trust from potential buyers • Engage the private sector and incorporate creative financing strategies
• Articulation with external markets will be limited by the ban on hilsa exports from Bangladesh, the lack of traceability in the supply chain, and no tradition of public-private partnerships	• Engage with Government to lift export ban • Fund in-depth supply chain analysis • Reduce Government ownership of the CTF as private sector engagement becomes more realistic over time
• Hilsa are a transboundary resource, lacking distinct boundaries, with distant and dispersed users, and so community-based management will be a challenge	• Coordination between Bangladesh, India, and Myanmar with the long-term goal of developing a regional CTF • A strong focus of grant-making should be on supporting self-monitoring and enforcement

Within the discussion of institutional sustainability, some consideration should be given to whether a PES needs to continue in perpetuity (or at least for a very long time), as often directed, or whether it may be more important to have a more short-term vision with an exit strategy. If the fundamental goal is behavioural change towards a profitable, sustainable fishery, in theory there should be no need to continue payments indefinitely. In particular, short-term payments can be useful if the activities being paid for will continue to be profitable once they have stopped (Wunder et al. 2008). However, PES schemes rarely have exit strategies, and more research is needed in order to be able to empirically determine when and how a PES scheme can be closed, i.e., at what point has a sustainable fishery been reached, such that payments do not have additional benefits? If a PES was determined to need a relatively short lifespan, a CTF could still play a useful role in brokering the partnerships that would be required to make it a transitional arrangement.

CTFs are not a panacea for PES, and there are many circumstances in which their creation will not be feasible or appropriate; when an intervention requires an immediate injection of funds, when there is limited government support, too much government interest, or lack of an appropriate legal framework. Furthermore, if poorly implemented, just like any other intervention, a CTF will have limited potential to advance PES. But if properly implemented, CTFs and PES can be mutually beneficial mechanisms. Building on the analysis in this chapter, a systematic assessment of the effectiveness and sustainability of conservation payments under a CTF framework would be valuable both to fisheries management and conservation science more broadly.

Chapter 8

Discussion

8.1 Overview

In tackling the problem of conservation payments in a developing-world fisheries context, three major research themes have emerged from this thesis. These themes constitute the major barriers to successful design and implementation of conservation payments in this context, and have implications for conservation science and fisheries management more broadly. The first theme is data poverty in fisheries and how to design, implement and evaluate payments in such circumstances; the second theme is trade-offs in conservation payments and the potential implications of poverty-targeting for ecological additionality; and the third is how to make conservation institutions such as PES and Conservation Trust Funds work in the real world. This thesis has contributed to progress in each of these three areas, and here I discuss each in turn. In doing so, I also provide policy recommendations for the management of hilsa in Bangladesh.

8.2 Data poverty

"It is better to be vaguely right than exactly wrong"

Carveth Read, *Logic: Inductive and Deductive* (1920)

Lack of data is one of the major challenges facing both fisheries managers and conservation practitioners, particularly in the developing world (Costello et al. 2012; Carruthers et al. 2014). To design or evaluate a conservation intervention, one first requires a clear baseline understanding of the social-ecological system, i.e., of ecological trends; of social, institutional, physical and economic factors; and of any interactions and feedbacks between them. To measure the true net benefit of the intervention, this frame of reference should incorporate not only baseline trends, but also counterfactual scenarios (Bull et al. 2014, 2015). In data-limited fisheries, uncertainties make it difficult to establish baselines and, particularly, to project counterfactuals. As Chapter 2 makes clear, this is a key hurdle to the translation of PES and other conservation payments from the terrestrial realm to marine and coastal environments. However, I have demonstrated that it is possible – even in complex, data-limited systems – to create useful frames of reference that allow counterfactual scenarios to be projected, and to assess the scope for, or confidence in, ecological additionality. This can in turn be used to guide fisheries management and, where appropriate, the design of conservation payment schemes.

In Chapter 3, I developed a qualitative frame of reference for the Bangladesh hilsa fishery, consisting of baseline trends and one positive and one negative counterfactual, using only understanding from the literature and limited secondary data. By comparing the positive and negative counterfactuals and critically evaluating the reliability of the sources of understanding, I identified key areas of uncertainty that could be used to guide hilsa management and future research; for instance, the sensitivity of hilsa to exploitation, the potential impacts of climate change on hilsa populations, and the biological basis for current hilsa management regulations. Building upon the No Net Loss conservation policy literature (Bull et al. 2014, 2015, 2016), this research is novel in its application of the frame of reference approach to fisheries, and contributes to the field of thought that places an emphasis on incorporating uncertainty into fisheries management decisions, even when modelling and manipulative experiments are impossible or unreliable (Davies 2015; Hilborn 2016).

Despite lagging behind other policy fields in its monitoring and evaluation efforts, conservation science increasingly highlights the importance of counterfactuals for attribution of impact, and thus the value of quasi-experimental evaluation (Ferraro & Pattanayak 2006). But given how rare it is for counterfactuals to be developed early in the design of conservation interventions, these kinds of evaluations are still difficult in practice (Gurney et al. 2014, 2015), and very seldom applied to payment schemes (Pagiola & Rios 2013; Clements & Milner-Gulland 2014). Although the counterfactuals developed in Chapter 3 provide invaluable context to guide future hilsa management, the limited availability of quantitative *ex-ante* baseline data means that counterfactual evaluation of either the current management package as a whole, or of the compensation scheme, is not yet possible. There are, however, numerous other evaluation approaches available, some of which are very new and have had limited application in conservation (Stern 2015). These approaches cannot be used to directly attribute impact to the interventions, but they can still be used for limited causal inference if the basis for causal claims is made explicit, and caveats are clearly expressed. In Chapter 5 I demonstrated the value of two of these approaches: one theory-based (relying on generative causality, i.e., the identification of mechanisms that explain effects) and one statistical (correlations between variables with controls for confounding variables).

Theory-based approaches to evaluation can be a powerful way to assess whether and how an intervention has potentially made or could make a difference, and thus provide an indication of confidence in the additionality of interventions in data-poor circumstances. There are various kinds of theory-based approaches emerging largely from the social and political sciences, but they all take the same general form as used in Chapter 5. Firstly, they involve mapping out a conceptual theory of change for the intervention (in other words, the mechanisms presumed to lead to the intended outcomes and their underlying assumptions). In this study, the theory of change was relatively straightforward to reconstruct using management plans and interviews, but a theory of change is best developed with the active participation of local stakeholders, particularly when projects have multiple strands, feedbacks and trade-offs. This process can be valuable even when there is a pre-existing theory of change available, because it opens the process up to different perspectives and insights (Woodhouse et al. 2015).

I examined the validity of this reconstructed theory of change by looking for empirical evidence to support or discredit its underlying assumptions, while at the same time considering the strength of this support and each assumption's level of importance in influencing the overall intended outcomes of hilsa management. As proposed by Befani & Mayne (2014), the assessment of strength of evidence would benefit from the application of principles and tests from process tracing, a robust and established probabilistic methodology developed for the analysis of historical events, which looks for evidence that increases confidence in the existence (or non-existence) of a theorised mechanism (Bennett 2008, 2010; Collier 2011; Mahoney 2012). Although my approach borrowed elements from this methodology, instead of simply measuring the strength or weakness of pieces of evidence in isolation, process tracing looks at the combination or accumulation of empirical observations and other contextual factors, using Bayesian logic to assess confidence in mechanisms within the theory of change (Barnett & Munslow 2014). Process tracing has not been used in conservation to date, but adaptation of the methodology for conservation impact evaluation might help to drive progress in reducing uncertainty around the contribution of interventions when counterfactual evaluations are not possible.

Together, Chapters 3 and 5 provide some insight into how and why current management might affect the hilsa fishery, and what might happen without institutional improvements. The frame of reference shows what might already be happening in the fishery, whereas the theory of change shows what kind of intervention might contribute to intended outcomes, given this context. The statistical and theory-based approaches in Chapter 5 are complementary; whereas the statistical analyses provide an indication that hilsa sanctuaries and compensation payments might have had some conservation impact, the theory of change provides insight into how individual components of management may be operating together to produce impact and why. Not only can these findings be used to guide future management decisions, they provide the basis for the development of a rigorous and long-term social and ecological monitoring and evaluation system, which should eventually allow attribution of outcomes. The theory-based evaluation involved the development of hypotheses and evaluation questions that should in turn help in the identification of controls and confounding factors in experimental and statistical designs, and in contextualising results. These methods hold promise for the design of conservation payment schemes, which thus far have largely been developed under the assumption of a connection between payments and outcomes (Pattanayak et al. 2010). The social impacts of these payments, in particular, have been poorly assessed, and the use of grounded theories of change in combination with counterfactual evaluations, where they are possible, should help to address this gap (Calvet-Mir et al. 2015; Woodhouse et al. 2015).

In a fisheries context, any monitoring and evaluation system would ideally include a reliable stock assessment. In the absence of such an assessment, management decisions are often based almost solely on biological norms from the literature, as is currently the case in the hilsa fishery. Yet, in these circumstances it is still possible to make more informed management decisions, whether by borrowing data from similar species or neighbouring populations (Smith et al. 2009; Honey et al. 2010), using local knowledge and perceptions (Fazey et al. 2006; Wilson et al. 2006), or through the application of novel data-poor methods for the assessment of stock status and sensitivity (Le Quesne & Jennings 2012).

The methods used in Chapter 4 add to the growing body of literature demonstrating that rapid assessments of stocks and their sensitivity to fishing can be conducted using minimal life history parameter data, which are much more available than survival and mortality estimates, or can be inferred from allometric relationships and related species (Beddington & Kirkwood 2005; Le Quesne & Jennings 2012; Zhou et al. 2012; Thorpe et al. 2015). Using these data, I was able to model the potential relative impacts of different harvesting strategies on hilsa population biomass, and found that the current hilsa management strategy is not likely to be cost-effective. Although harvesting of juveniles is widely assumed to contribute to stock depletion, I showed that, for hilsa, exploitation rate is more influential than size selectivity. It follows, therefore, that targeting *jatka* fishers may not be a cost-effective strategy for a payment scheme. Moreover, I showed that hilsa are highly resilient to exploitation, which indicates that limited resources might be better spent targeting the physical drivers of change in the hilsa fishery – particularly pollution and the water diversion activities that are blocking migratory pathways (Chapter 5) – than fishing itself. These findings highlight the importance of grounding conservation payments in robust biological understanding. They are also likely to be transferable to other forage fish with similar life histories, such as American shad and herring.

Chapter 4 also demonstrates the potential for management recommendations to be made based on creative ways of using and combining unconventional data, when nothing else is available. Despite the numerous assumptions made in this study, the combined use of fishers' knowledge and life history parameters was an important first step to understanding the distribution of household-level impacts of fishing on hilsa. Future studies could be improved if, for instance, actual landings data were available, but the value of fishers' ecological knowledge – particularly that of small-scale fishers – should not be underestimated (Johannes 2000). Fishers' knowledge has been used in fisheries assessments (Neis et al. 1999; Bender et al. 2013), in the design of conservation and fisheries management interventions (Drew 2005; Wilson et al. 2006), and in a number of studies that have demonstrated complementarity between fisher knowledge and conventional scientific knowledge (Silvano et al. 2008; Silvano & Valbo-Jørgensen 2008; Jackson et al. 2014; Gaspare et al. 2015).

Expert and experiential knowledge (from fishers and a range of other individuals) and fishers' perceptions of ecological trends also underpinned much of the theory-based and statistical evaluation of additionality in Chapter 5, and the assessment of compensation scheme sustainability in Chapter 7. Local perceptions (which can be based on, but are distinct from, knowledge) can provide insights on social impacts of conservation, ecological outcomes of conservation, legitimacy of conservation governance, and acceptability of management, which ultimately determine conservation success (Bennett 2016). As demonstrated in Chapter 5, perceptions can rarely be used to determine causality; even experimental and quasi-experimental studies of perceptions are limited by their subjectivity (Gurney et al. 2014). Nonetheless, well-designed studies of perceptions provide a rapid way to establish baselines or understand effectiveness when quantitative experimental and quasi-experimental evaluations are not possible, and can provide vital support even when they are. And yet, as Bennett (2016) laments, this kind of evidence is rarely incorporated into monitoring and evaluation programmes. Research on perceptions deserves more attention as one way to improve adaptive, evidence-based management.

Of course, there are limitations to all knowledge derived from personal experience. Perceptions can vary widely between different groups and individuals, and there is a risk that responses will reflect what the individual thinks that they should say or what they think the researcher wants to hear (Fazey et al. 2006; Newing 2011). For instance, strategic responses are common when individuals perceive a risk of sanctions or a potential benefit. Fishers' knowledge should therefore be validated by triangulation with information obtained from different respondents or different methods of questioning, and preferably by using it in combination with other types of data (Daw et al. 2011b; Newing 2011). Measures should also be taken to reduce the risk of strategic bias; for example, by using specialised indirect questioning techniques (Nuno & St John 2015) and by building trust in the independence of the research team (Bernard 2011).

8.3 Trade-offs in conservation payments

"For everything you have missed, you have gained something else, and for everything you gain, you lose something else"

Ralph Waldo Emerson

Conservation interventions increasingly seek to achieve both conservation impacts and positive social impacts, while minimising economic costs. Yet, when interventions are designed to meet multiple objectives, complex trade-offs are usually encountered. Win-win or triple-win solutions may be possible when objectives are positively correlated, or independent of one another, but improving outcomes in one dimension will usually compromise outcomes in another (Ferraro & Hanauer 2010; McShane et al. 2011; Salafsky 2011). The new conservation debate thus challenges conservationists to identify and openly examine these trade-offs, so that stakeholders may discuss and negotiate hard choices in decision-making (Minteer & Miller 2011; Hirsch et al. 2011). Trade-offs are a key element of the ongoing debate around the potential for PES to contribute to poverty alleviation (Pagiola et al. 2005; Bulte et al. 2008; Wunder 2008). This research contributes to the debate by tackling the question of whether explicit poverty-targeting, commonly used in public payment schemes, can generate win-win outcomes.

Together, Chapters 4 and 6 demonstrate how trade-offs can limit the potential for win-win outcomes, especially when these trade-offs are not acknowledged. Currently, the compensation scheme for hilsa fishers in Bangladesh is failing to effectively target for either its social or ecological objectives; compensation allocation is spatially (and thus probably politically) driven, with limited association to indicators of household fishing dependence, poverty, vulnerability (Chapter 6) or *jatka* fishing activities (Chapter 4). Having established this, I went on in Chapter 6 to consider the implications of two potential alternative targeting strategies. I considered compensating all *jatka* fishers to be the least favourable of these due to the general vagueness of the term '*jatka* fisher', the related challenges of identifying who should receive compensation, and potential inequities imposed by the fact that some individuals who target *jatka* are reported to be seasonal or occasional fishers and therefore not highly dependent on hilsa. Moreover, the finding that potential household impact on hilsa population biomass is driven more by exploitation rate than size selectivity (Chapter 4) indicates that this targeting strategy would in fact have low potential for ecological additionality, given the lack of correlation between *jatka* fishing and high catch volumes.

Instead, I recommended targeting the households who are most dependent on hilsa fishing as a more practicable and equitable strategy, which should be more cost-effective both socially and ecologically (Chapter 6). But, again, in light of Chapter 4, the potential for this strategy to have ecological additionality seems low. Although *jatka* households were found in Chapter 6 to have a marginally higher dependence on fishing than non-*jatka* households, in Chapter 4 I found that the households who are most dependent on hilsa fishing, and hence most in need of social support, are not the same ones who catch the greatest volumes of hilsa, and hence have the greatest potential for ecological additionality, indicating a strong trade-off between social and ecological objectives of the scheme. Even with implementation of the institutional changes recommended in Chapters 6 and 7, the scheme's ecological additionality has potential to be undermined by its social targeting goals, and vice versa.

Although it is often described as a PES – and one more concerned with hilsa conservation than poverty alleviation (Islam et al. 2016; Porras et al. 2016) – currently, the compensation scheme more resembles a social safeguard compensating for costs incurred by fishing bans than a conservation payment. Not only is it based on the pre-existing Vulnerable Group Feeding (VGF) and (to a lesser extent) alternative livelihood support programmes, it lacks enforcement of conditionality. The VGF was introduced as a short-term form of relief for households considered to be at high risk of hunger, particularly natural disaster victims (Uraguchi 2011). Key informant interviews suggested that the top-down approach to distribution of compensation to fishers has limited the potential for it to be viewed as anything more than relief (Chapter 7). Moreover, attempting to reframe this programme by introducing conditionality on fishing behaviour – even if it were effectively enforced – could be viewed as problematic due to its potential to exclude very vulnerable households. Making relief available only to hilsa or *jatka* fishers could also generate a perverse incentive to participate in these activities. Similar concerns have been raised over a scheme in the Brazilian Amazon that distributes subsidies to artisanal fishers as compensation for a closed fishing season. Corrêa et al. (2014) demonstrated that this cash compensation has actually contributed to an increase in numbers of fishers.

It is clear that the compensation scheme needs a clearer and more focused objective to guide implementation. The lack of synergy between the social and ecological targeting parameters could even be taken as evidence of a need to separate the issues of hilsa conservation and poverty alleviation. PES is widely advocated for its economic efficiency – a conceptualisation that focuses on the cost-effectiveness of reducing ecological impacts, i.e., by targeting the most effective providers of ES (in this context, fishers with the greatest ecological impact on hilsa) at the lowest cost (Wünscher et al. 2008). There is therefore a large body of literature investigating optimal payment design by trying to identify geographical locations or individuals with synergies between high potential additionality, low opportunity costs and high poverty levels (Alix-Garcia et al. 2008; Gauvin et al. 2010; Zhang & Pagiola 2011; Jindal et al. 2013). Some of these investigations have found good synergies between these parameters, but trade-offs between poverty alleviation and cost-effectiveness are often high. Many authors have thus warned against poverty alleviation becoming a primary goal of interventions with a conservation objective (Pagiola et al. 2005; Wunder 2008), and even called for it to be addressed separately (Kinzig et al. 2011).

The focus of PES on efficiency and subsequent quests for optimal solutions has been criticised by others (Kroeger 2013; Pascual et al. 2014; Kolinjivadi et al. 2015). Optimisation studies frequently ignore uncertainty by using unidimensional conceptualisations and quantitative measures of poverty, and using opportunity costs as a bench-mark for efficiency or cost-effectiveness, without full consideration of the underlying dynamics at a household level (Alix-Garcia et al. 2008; Gauvin et al. 2010; Zhang & Pagiola 2011). In our household survey, we collected data on opportunity costs by asking respondents to estimate a) how much household income they lose as a direct result of to the fishing ban (and any extra income they get during that period from activities they would not have done outside of the fishing ban); and b) their monetary willingness to accept compensation. These data could have been incorporated into a trade-off analysis with fishing dependence and potential ecological impact. I decided not to use them when preliminary analysis showed little convergence between the two proxies of opportunity cost, and high heterogeneity; these kinds of questions are very leading and the proxies they generate are notoriously inaccurate (Gregersen et al. 2010). Moreover, they should not be used in isolation from the socioeconomic processes driving them (Adams et al. 2010; Kolinjivadi et al. 2015).

Numerous researchers have called for a more explicit consideration of equity in targeting payments (Corbera et al. 2007; Kroeger 2013; Pascual et al. 2014). Equity has multiple dimensions, the relative importance of which are context dependent: procedural legitimacy (i.e., degree of participation in decision-making); the distribution of costs and benefits of conservation; and the recognition of stakeholder rights, norms and values (Pascual et al. 2014). It is often assumed that increased equity comes at the cost of ecologically optimal solutions, but in fact this complex relationship is rarely established; the degree of equity in an intervention can affect social and ecological outcomes in myriad ways (Pascual et al. 2010). As a result, optimisation studies in the PES literature and wider spatial planning literature have begun to incorporate dimensions of equity (Gross-Camp et al. 2012; Halpern et al. 2013; Narloch et al. 2013). Kolinjivadi et al. (2015), for instance, emphasise that targeting efforts should not only identify trade-offs, but also assess the extent to which those trade-offs are socially legitimate. I examined perceptions of fairness of compensation distribution in the hilsa fishery, and found the majority of respondents reported perceptions of unfair distribution, indicating a lack of social acceptance that is probably limiting conservation outcomes (Fehr & Falk 2002; Sommerville et al. 2010a). They also supported targeting the most dependent fishing households, which is evidence that a more socially-focused targeting strategy would be more socially acceptable. However, I was unable to address the other dimensions of equity – a gap that needs filling.

With this in mind, I would argue that the potential for hilsa conservation payments to concurrently provide social and ecological benefits requires further investigation. The existence of trade-offs should not prevent conservation payments from having positive social impacts, if they are well designed and implemented and the trade-offs are made clear from the outset (Clements & Milner-Gulland 2014; Bremer et al. 2014; Porras et al. 2016). But when trade-offs between social and ecological goals are strong, explicit poverty-targeting mechanisms may not be appropriate (Wunder et al. 2008; Wunder 2008; Ingram et al. 2014). Instead of attempting to integrate social and ecological goals, it can be preferable to focus on conservation goals, which can be a means to achieving social goals in the long term (Salafsky 2011). If conservation payments are to help fisheries contribute to poverty alleviation on a global scale, perhaps the focus should be on scaling up ecologically additional schemes rather than on participation of the poor. Ultimately, more quantitative empirical research investigating how payment schemes may account for complex trade-offs in data-poor circumstances is needed.

One way to address the ineffective social targeting in the hilsa fishery might be to extend coverage of conditional compensation to all households within the compensation area (or the area deemed to have the greatest potential for ecological additionality). Research on targeting effectiveness of social safeguards to mitigate negative impacts of REDD+ in Madagascar has highlighted the challenges facing social assessments of households in very poor, remote communities, where assessors can be heavily influenced by local elites, and where households can be reluctant to reveal their dependence on a resource (Poudyal et al. 2016). They concluded that in systems like this, and where the majority of households are very poor and likely to be incurring costs, compensating all could be the optimal approach, avoiding the costs of ineffective social assessments. The situations are slightly different in that REDD+ social safeguards are designed to compensate local communities for access restrictions from which they do not clearly benefit, whereas conditional compensation for hilsa fishers is designed to incentivise compliance with fishing bans from which they should benefit in the long term through a more sustainable and productive fishery. The literature on targeting resource transfers in development also highlights the risks of dependence and perverse outcomes that can arise from 'hand-outs' (Slater 2011), which, in a system where conditionality of compensation is poorly enforced, would be high. Nevertheless, an assessment of the relative costs and benefits of removing the social targeting vs. a more effective social targeting strategy in the hilsa fishery – which would involve extensive social surveys and may not be achievable under current governance – might be worthwhile.

The conservation potential of either strategy would still ultimately depend upon whether the regulations and thus the compensation area were appropriately spatially placed, and Chapters 4 and 5 indicate that they may not be. Fig. 4.12a clearly illustrates that, at the district level, the households with greatest potential impact on hilsa are living outside of the area where the compensation scheme currently operates, in Cox's Bazar. Fishing regulations (focused on effort reduction) and any appropriate concurrent payments might therefore be better targeted towards marine fishers than river fishers.

8.4 Making conservation institutions work in the real world

"The devil is in the detail"

Anon

This research highlights just how difficult it can be to put abstract ideas into conservation practice. As new tools such as conservation payments and Conservation Trust Funds (CTFs) emerge it is wrong to assume that they are panaceas for complex problems (Ostrom 2007) – a mistake that has bred a cycle of conservation fads in recent decades (Redford et al. 2013). In fact, success fundamentally depends on how well a conservation institution fits the environmental, institutional, social and economic contexts in which it operates (Corbera et al. 2009; Muradian et al. 2013; Sarkki & Karjalainen 2015). A crucial challenge in conservation science is therefore to understand the conditions under which a particular conservation institution can effectively govern a social-ecological system.

Together, Chapters 2 and 7 identify conditions under which conservation payments are most likely to be successful in developing-world fisheries. In Chapter 2, I identified the preconditions that should help to ensure successful implementation of a PES scheme in these fisheries. Although I went on to demonstrate how rare it is that such a fishery will fulfil these preconditions *a priori*, the key here is capacity for improvement. In Chapter 7, I investigated one way of building this capacity: a CTF framework. Further to their financial stability and independent intermediary roles, CTFs have the potential to help build capacity for monitoring and enforcement, which should thereby allow conditionality and additionality to be established. Yet, my investigation of how a CTF might help to catalyse the development of a sustainable PES in Bangladesh highlighted how context-specific this potential is; there are circumstances when the wider institutional context can limit best practice implementation, in which case a CTF might not provide the expected benefits. For instance, too much or too little government ownership of a CTF can limit effective governance, or an intervention might require funds more immediately than the CTF can generate them. Capitalisation of a CTF can be particularly difficult when best practice governance standards are not followed and when countries lack a tradition of public-private partnerships (Bladon et al. 2014a). I showed in Chapter 2 how PES could guide private sector investment in fisheries improvement, but the potential role of CTFs in realising this will depend on their level of private sector engagement (Chapter 7). The role of the private sector in marine conservation is an under-explored but promising route to closing the funding gap in developing-world fisheries, and one which deserves further research attention in the context of PES and CTFs (Bos et al. 2015).

A major challenge for conservation payments in developing-world fisheries is the design and implementation of payments that actually incentivise behavioural change on the ground. Conditionality is an essential component of incentive-based approaches, and in systems facing monitoring challenges this conditionality will usually have to be based on behaviour rather than outcomes (Sommerville et al. 2009; Bladon et al. 2014b). When payments are used in tandem with fishing regulations, as is the case in the hilsa fishery, the payments should have some level of conditionality on compliance with the regulations. But, as demonstrated in Chapter 2, conditionality can be very difficult to establish in marine and coastal systems in developing countries, where monitoring and enforcement for compliance can have high transactions costs. In the hilsa fishery, weak monitoring and enforcement by government agencies has compromised the potential effectiveness of the carrot-and-stick approach; the compensation payments might be offsetting short-term costs, but they are rarely motivating compliance (Chapter 5).

The focus of conservation payments should not be, in these circumstances, on establishing strict top-down conditionality, but on building community-level institutions for collective action

(Noordwijk & Leimona 2010; Kerr et al. 2014). In a collective scheme, individuals can work together to agree on the conditions of payment and participate in monitoring and enforcing conditionality, while payments can be designed to help cover the costs of doing so (Clements et al. 2010; Sommerville et al. 2010b). Individual payments can still be conditional on collective action, but collective payments can be more sustainable in socially cohesive groups due to lower transaction costs and the kinds of institutions that are created to manage them (Clements et al. 2010; Travers 2014; Midler et al. 2015). Yet, community-level institutions should not operate in isolation; Chapters 2 and 7 clearly establish the necessity for conservation payments in a developing-world fishery context to be under a hybrid and multi-level form of governance. Building on previous suggestions that PES and co-management institutions could be mutually beneficial (Begossi 2014; Sarkki & Karjalainen 2015), more research at the interface of co-management, PES and CTF institutions would support the effective implementation of collective payments in fisheries management.

Collective or not, it is important to remember that conservation payments do not operate in a vacuum; individuals' decisions are influenced by a whole range of other formal and informal rules and perceptions (Vatn 2010; Sommerville et al. 2010b; Bennett 2016). Without a thorough understanding of the wider social and institutional context in which a payment operates, there is a risk that payments will not incentivise compliance (Hayes et al. 2015). Indeed Vatn (2010) suggested that more research is needed to better understand the relationships and perceptions behind payments as incentives and payments as compensation. For instance, payments can crowd out pre-existing intrinsic motivations for conservation, or lead to perceptions that the intervention is unfair, which can affect compliance (Fehr & Falk 2002; Sommerville et al. 2010a). In Bengali culture there is a custom of not eating hilsa during the presumed breeding season (Sharma et al. 2012; see Chapter 3), but I found no other evidence of pre-existing social norms for hilsa conservation, and so the risk of crowding out seems minimal in the hilsa fishery.

The format of a conservation payment – for instance, in kind vs. monetary – must also be appropriate for the social and institutional context. Although in-kind incentives can face greater challenges in terms of conditionality (Goldman-Benner et al. 2012), they have become more common with the rise in collective PES and are thought to be less likely to interfere with intrinsic motivations (Kerr et al. 2014). Food-based payments, however, are rare in conservation. In Chapter 5, I found evidence to suggest that both the rice compensation and alternative livelihood support schemes are limited in their potential to incentivise compliance in Bangladesh, partly due to limited institutional fit. By providing a short-term boost to food security, the rice compensation scheme does little to lift fishers from the cycles of debt in which they are trapped, and for which they still need to pay interest during the fishing bans (Chapter 5). It seems that, for many of these fishers, the costs of the management regime are too high for rice compensation to incentivise compliance, even if conditionality were enforced. Local calls have been made for increased coverage of livelihood-focused payments, with careful stakeholder consultation, that should instead help fishers to build the adaptive capacity to escape debt (Chapter 5). Yet, there is evidence to caution against the use of alternative livelihood support to incentivise sustainable behaviour, since it does not necessarily reduce the need or desire to exploit resources (Wright et al. 2016). Instead, a conditional loan or cash transfer could help to lower opportunity costs and provide an entry point into the conservation payment (Porras et al. 2016). As suggested in Chapter 7, this could take the form of CTF microcredit with repayments tailored to the timings of the fishing bans (Mohammed et al. 2014).

Beyond incentivising compliance, unintended feedbacks triggered by human reactions to conservation payments can result in poor conservation outcomes (Larrosa et al. 2016). A major risk of the carrot-and-stick approach is perverse incentives, particularly when enforcement of regulations is very weak (Wunder 2007; Corrêa et al. 2014). I did not specifically look for evidence of perverse incentives in the hilsa fishery, but there is evidence of unintended feedbacks that could be driving entry into the hilsa fishery (Chapter 5). Firstly, since only hilsa fishers are eligible for compensation, Islam et al. (2016) observed that other fishing households who are still affected by the fishing bans are entering the hilsa fishery in order to receive compensation. Secondly, the distribution of rice compensation causes a decline in local rice prices and demand during the fishing bans, while the fishing bans drive an increase in demand for informal loans, an increase in interest rates and – as fishers seek alternative income from casual labour – a decline in local labour wages (Mohammed et al. 2014; Uraguchi & Mohammed 2016). Since *jatka* fishing is often an opportunistic activity that individuals switch to when their income is low (Haldar & Ali 2014), it is probable that these impacts on the local rice, microfinance and labour markets are actually leading to perverse conservation outcomes. This risk deserves further research attention, starting with the integration of these potential unintended feedbacks into a theory of change (Chapter 5), and the development of an improved vessel monitoring system (Chapter 3).

Finally, I emphasise the importance of flexibility in conservation payment design (Chapter 2). If one focuses too much on the market rationale that underpins much of the PES literature (Engel et al. 2008), there is a danger of making overly simplistic assumptions about compensation payments. Market-based framing has led numerous researchers to reject PES for its commodification and neoliberalisation of nature (Büscher 2012; Dhandapani 2015). Sceptics view market-based approaches as inappropriate solutions that bind conservation to the global corporate interests that undermine it (Büscher et al. 2012) and argue that the commodification of nature fails to recognise the complexity of social-ecological systems (Kosoy & Corbera 2010). But the majority of real-world conservation payments do not follow a free market rationale at all. Like the compensation scheme in Bangladesh, most are subsidy-like government payments, not based on monetary valuation of nature but on opportunity costs, and which rarely price or trade ES according to the rules of supply or demand (Sommerville et al. 2009; Muradian et al. 2013; Hahn et al. 2015; Vatn 2015). Going forward, we must recognise the distinction between incentives and market transactions, and focus the conservation payment research agenda on adapting the theoretical foundations of PES for more socially-informed applications in varied contexts (Van Hecken et al. 2015).

8.5 Conclusions

Conservation payments offer an opportunity to address social and ecological failings in fisheries management by incentivising behavioural change. Weak institutions and data limitations pose significant challenges to payment design and implementation in the developing world, but these challenges are not insurmountable. It is important to recognise that there are methods available for the design and evaluation of conservation interventions in data-limited systems, and that it is better to use these than to do nothing at all. A flexible approach to conservation payment design and careful integration with other institutions should allow the institutional capacity-building that will be required in most developing-world fisheries. Trade-offs between social and ecological objectives can be strong, so care should be taken not to lose sight of conservation in pursuit of poverty alleviation. There is, however, a limit to the role that conservation payments can play in fisheries management. There will be situations where conservation payments are inappropriate, infeasible or less cost-effective than a regulatory approach. With more empirical research there will come more opportunity to make decisions based on experience rather than ideology.

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Appendix A: Supplementary material for Chapter 3

Table A.1: Fishery production (metric tonnes) in Bangladesh from 1983-84 to 2011-12 (DoF 2014a).

Year	Inland capture	Marine capture	Total capture	Aquaculture	Country Total	Inland hilsa	Marine hilsa	Total hilsa
1983-84	471595	164882	636477	117025	753502	90082	56000	146082
1984-85	462605	187563	650168	123811	773979	73328	71050	144438
1985-86	441799	207401	649200	144723	793923	94794	96294	191091
1986-87	431006	217579	648585	166100	814685	91167	103814	194981
1987-88	423598	227582	651180	175925	827105	78551	104950	183501
1988-89	424140	233281	657421	183505	840926	81641	110311	191952
1989-90	423872	239063	662935	192592	855527	112408	113943	226351
1990-91	443404	241538	684942	210993	895935	66809	115358	182167
1991-92	479742	245474	725216	226863	952079	68356	120106	188462
1992-93	532419	250492	782911	237743	1020654	74715	123115	197830
1993-94	573376	253044	826420	264190	1090610	71370	121161	192531
1994-95	591145	264650	855795	317073	1172868	84420	129115	213535
1995-96	609151	269702	878853	379087	1257940	80625	126660	207285
1996-97	599900	274704	874604	485864	1360468	83230	131204	214434
1997-98	615949	272818	888767	574812	1463579	81634	124105	205739
1998-99	649418	309797	959215	593202	1552417	73809	140710	214519
1999-00	670465	333799	1004264	657120	1661384	79165	140367	219532
2000-01	688920	379497	1068417	712640	1781057	75060	154654	229714
2001-02	688435	415420	1103855	786604	1890459	68250	152343	220593
2002-03	709333	431908	1141241	856956	1998179	62944	136088	199032
2003-04	732067	455207	1187274	914752	2102026	71001	184838	255839
2004-05	859269	474597	1333866	882091	2215957	77499	198363	275862
2005-06	956686	479810	1436496	892049	2328545	78273	198850	277123
2006-07	1006761	487438	1494199	945812	2440011	82445	196744	279189
2007-08	1060181	497573	1557754	1005542	2563296	89900	200100	290000
2008-09	1123925	514644	1638569	1062801	2701370	95970	202951	298921
2009-10	1029937	517282	1547219	1351979	2899198	115179	198574	313753
2010-11	1054585	546333	1054585	1460769	3061687	114520	225325	339845
2011-12	957095	578620	1139388	1726067	3261782	114475	232037	346512

Table A.2: Boats operated per year and average catch (metric tonnes) per boat per year in the marine sector in Bangladesh from 1983-84 to 2005-06 (Mome & Arnason 2007).

Year	Number of boats			Average catch/boat/year (mt)		
	Mechanised	Non-mechanised	Total	Mechanised	Non-mechanised	Total
1983-84	3347	-	3347	16.70	-	16.70
1984-85	3000	-	3000	23.70	-	23.70
1985-86	2887	3802	6682	30.60	2.08	14.40
1986-87	2887	3800	6680	32.90	2.36	15.50
1987-88	2882	3509	6389	31.90	3.76	16.40
1988-89	2880	3509	6389	33.00	4.37	17.30
1989-90	2880	3509	6389	33.10	5.32	17.80
1990-91	2880	3509	6389	33.90	5.06	18.10
1991-92	2880	3509	6389	35.40	3.14	18.80
1992-93	2880	3509	6389	36.50	5.13	19.30
1993-94	2880	3509	6389	36.10	4.94	19.00
1994-95	2880	3509	6389	38.70	5.03	20.20
1995-96	2880	3509	6389	37.90	4.95	19.80
1996-97	2880	3509	6389	39.90	4.64	20.50
1997-98	2880	3509	6389	38.30	3.72	1.40
1998-99	2880	3509	6389	42.30	5.35	22.00
1999-00	18982	7177	26169	6.30	2.94	5.40
2000-01	18982	6377	25369	6.91	3.67	6.00
2001-02	18982	6377	25369	6.93	3.24	5.90
2002-03	18982	6377	25369	6.02	3.42	5.36
2003-04	18982	6377	25369	8.30	4.28	7.29
2004-05	18982	6377	25369	9.00	4.33	7.82
2005-06	18982	6377	25369	9.00	4.48	7.84

A.1 Administrative hierarchy

Bangladesh comprises eight administrative divisions, which are in turn subdivided into sixty-four districts. Each district comprises sub-districts called *upazilas*, which are further divided into unions, and finally wards, which contain a number of villages (Ahmed et al. 2010). District and division level administration is led by central Government officials. Urban local governance operates through six city corporations (Dhaka, Chittagong, Rajshahi, Khulna, Barisal, and Sylhet) each led by a Mayor. Unions and *upazilas* in rural areas are governed by democratically elected local councils, called *parishads* (Fig. A.2). Union *parishads* are made up of representatives from each village and a chairman. *Upazila parishads* include union *parishad* chairmen, mayors, an elected chairman and two vice chairmen. Central Government bureaucrats also have influence at the *upazila* level via the *upazila nirbahi* officers and ministry representatives, and central Government bureaucracy is reported to impede local administration (Ahsan 2010). Local village leaders tend to make community decisions.

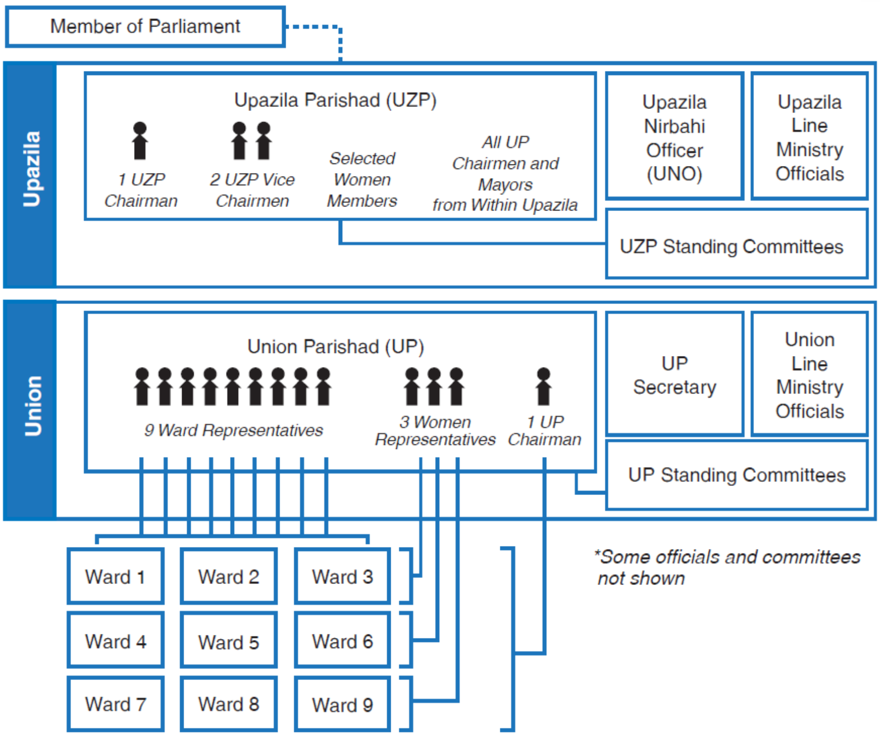

Figure A.1: Key elements of rural local administration. Reproduced with permission from Christensen et al. (2012).

A.2 Marketing system

Alam et al. (2012b) identified 4-6 main intermediaries in the hilsa supply chain, depending on the type of market. The marketing system varies from place to place, as do the local names of the intermediaries, but its basic three-level structure is described in Fig. A.3. The key players are the *aratdars*, who connect buyers and sellers of fish by handling the auctioning process at wholesale markets, taking a commission of up to 5 per cent. They often invest in fishing activities by giving loans to fishers, wholesalers/suppliers (known locally as *paikars* or *beparis*) and retailers. They usually have their own storage facilities (*arat*) and may employ staff. There are two types of *aratdar*: those who collect fish from local *paikars*, mobile collectors (*farias*), *mohajons*, or directly from fishers, and sell in local markets; and those who operate second auctions in urban areas or more distant markets. *Paikars/beparis* can play the role of local suppliers at the primary market level or as wholesalers supplying fish from local and distant wholesale markets to retailers, who then sell on to consumers at fish markets and as street vendors. Only LC (letter of credit) *paikars* are licensed to export hilsa overseas.

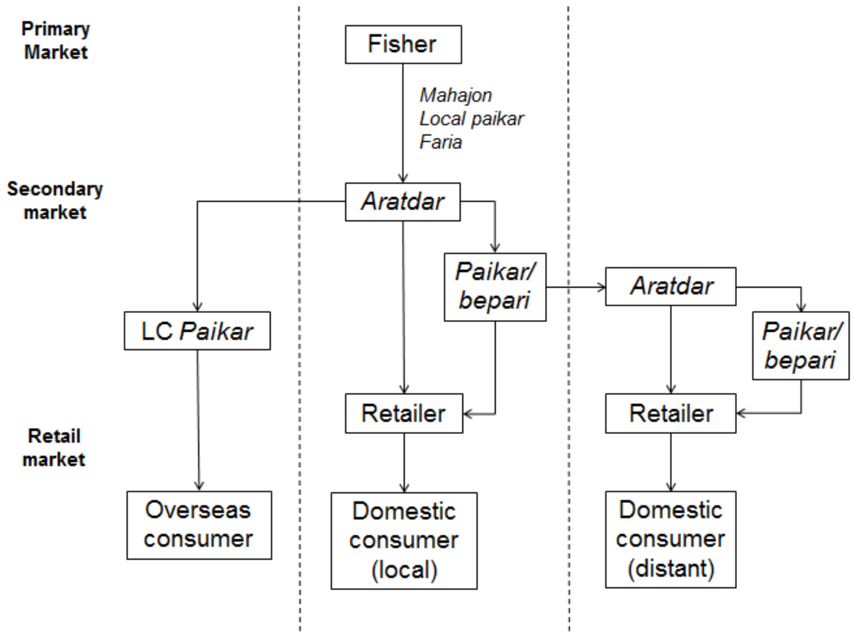

Figure A.2: A simplified representation of the domestic and foreign market chains for hilsa in Bangladesh (Alam et al. 2012b). Three market levels exist. LC = letter credit. The terms *paikar, bepari, faria* have different meanings in different regions.

This marketing system requires icing and transportation and therefore employs a large number of day labourers, including women and children (Ahmed 2007). The intermediaries have complete control of the system, and with little to no Government regulation it has been criticised for exploitation of fishers (Islam 2003; Haque 2011; Alam et al. 2012b). Fishers lack market information and bargaining power, particularly in remote areas with poor transport links, where they have no choice but to sell to intermediaries (Ali et al. 2010). Furthermore, the presence of so many players in the supply chain raises the retail price and the costs to the fisher. Supply chain analysis conducted on hilsa caught in the coastal Patuakhali district indicate that marketing profits received by intermediaries are relatively high, with only 55 per cent of retail price going to the fishers (Ahmed 2007). Another analysis conducted in Chandpur district found fishers receiving 31 per cent of retail price, with the major share of profit going to the *mahajons* (Alam et al. 2012b). *Dadandars* (*aratdars* and *mahajons*) in particular play a key role in determining market prices, through the extension of credit (*dadan*) which requires fishers to sell their catch back to/through them below the market price (Kleih et al. 2003). Although the *dadan* is interest free, this price differential is an informal equivalent of interest.

Block ice is used for preservation on trawlers and mechanised artisanal boats which fish for more than one day (Kleih et al. 2003). If the transportation time from primary market to retail market is in excess of six hours, it is usually iced (Ahmed 2007). Alam et al. (2012a) found post-harvest loss to be minimal, except in peak fishing season when supply exceeds the availability of ice. Hilsa are preferred fresh, but dry and wet salting and salt-fermentation are used as methods of long-term preservation, particularly during peak fishing season (Alam 2010). It cannot be sun-dried due to its high lipid content. Low value hilsa may be frozen and sold more cheaply on the domestic market. High value hilsa for the export market are frozen whole using semi-IFQ (individual quick freezing), sometimes in shrimp processing plants (Alam et al. 2012).

A.3 Analysis of climate data

Based upon data collected at 5 meteorological stations within the study area in Bangladesh (Barisal, Bhola, Chandpur, Patuakhali and Cox's Bazar), I calculated mean total monthly temperature and mean monthly rainfall for the period of 1983 to 2014 (Fig. A.1). This period was chosen to coincide with the period over which catch monitoring of hilsa has taken place in Bangladesh. These data are not necessarily representative of the entire study area or of the entire country – rainfall and temperature data were variable between stations. However the trends in these data were consistent between stations.

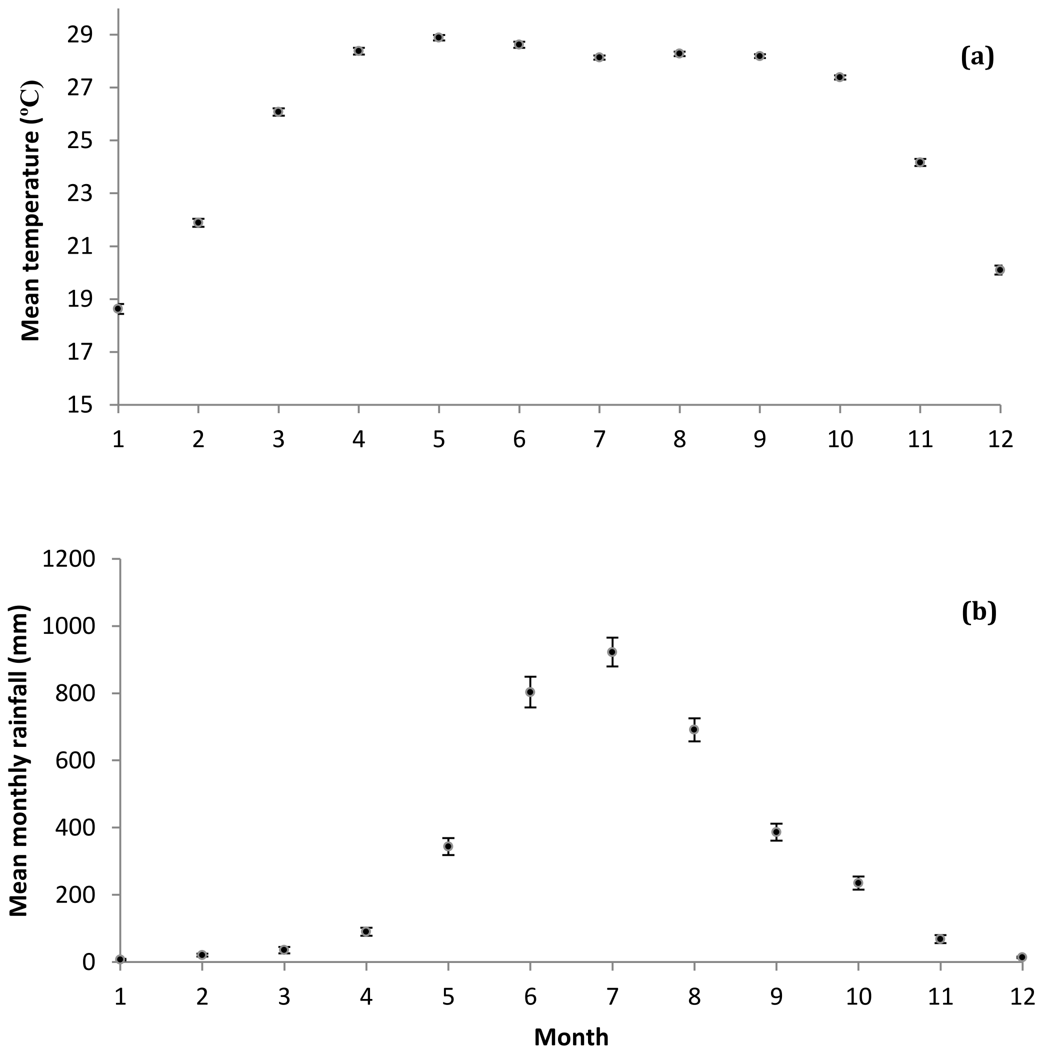

Figure A.3: Inter-annual variation in **(a)** mean temperature and **(b)** mean total monthly rainfall from 1983-2014 across 5 meteorological stations in the study area. Error bars show 95% confidence intervals. Source: Bangladesh Meteorological Department (2014).

Appendix A references

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Appendix B: Supplementary material for Chapter 4

B.1 Household survey methods

This household survey was conducted as part of a project led by the International Institute for Environment and Development (IIED) in collaboration with the Bangladesh Centre for Advanced Studies (BCAS). I made substantial contributions to the survey design and participated in training of the enumerators and piloting of the survey, but final design decisions were made as a team and interviews were conducted by enumerators from BCAS[38].

900 households were interviewed across 23 villages in eight districts of southern Bangladesh (lower Meghna River region, mid-Padma near Rajbari and coastal/marine area in Cox's Bazar district; Fig. B.1). Interviews were conducted between May and October 2014. 800 of these households were selected from clusters of villages in the six districts where the compensation scheme operates (Chandpur, Laxmipur, Bhola, Patuakhali, Barisal and Barguna; Fig. B.2). 600 of these households were selected from hilsa sanctuary areas (Chandpur, Laxmipur, Bhola, Patuakhali districts) and 200 from districts outside sanctuary areas (Barisal and Barguna). The remaining 100 households were selected from one inland district (Rajbari) and one coastal district (Cox's Bazar) where the compensation scheme does not operate (controls).

[38] Where possible, the same team of three enumerators went to all villages and divided up the selected households. In three villages (Rayrabad and Charkachopia in Bhola district and Baushia in Barisal district) an additional enumerator conducted some of the questionnaires.

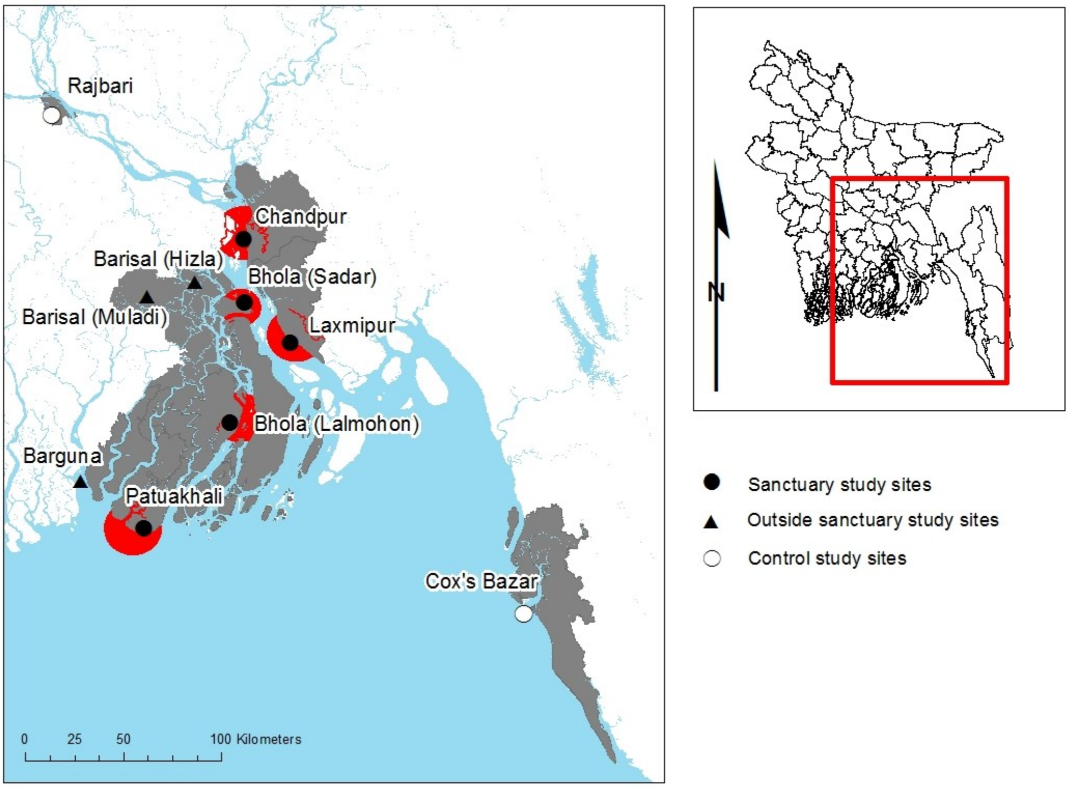

Figure B.1: Map of study area, showing study site districts (grey) in relation to sanctuary sites (red). Control study sites are outside of the area where the compensation scheme operates. Each study site represents the approximate location of a cluster of surveyed villages, denoted by the relevant district name (precise village coordinates were not available). In Barisal and Bhola districts two village clusters were sampled and can be distinguished by the sub-district names (in brackets); in the other districts just one village cluster was sampled.

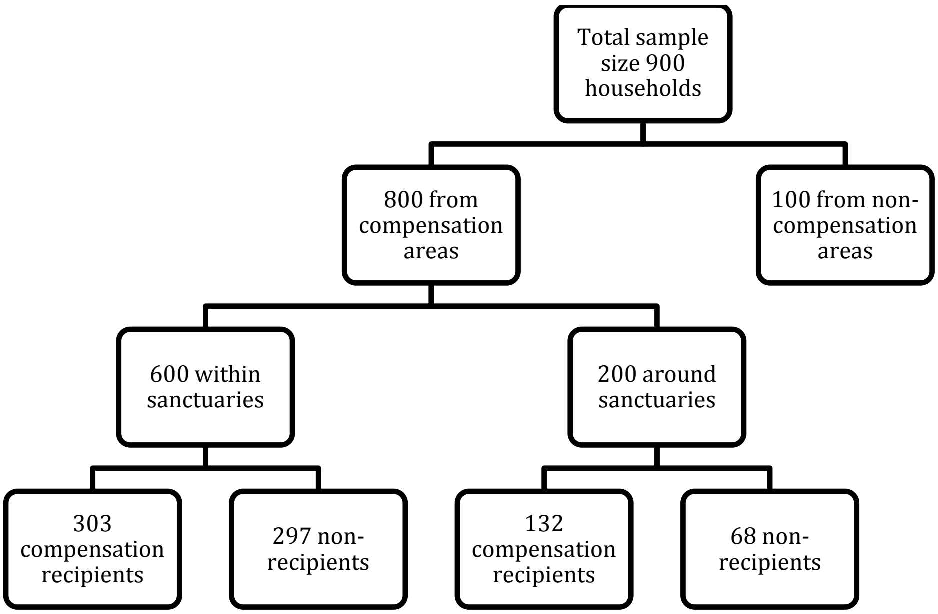

Figure B.2: Household survey sampling design.

Households were selected from each district through stratified random sampling (see Table B.1). Inside the compensation area, 150 households were sampled from each district. Due to resource constraints, sample sizes were smaller outside sanctuary areas (125 households in Barisal district and 75 in Barguna district) and outside the compensation areas (50 households in each district). We aimed to balance the proportions of recipients and non-recipients interviewed within the compensation areas; 435 (54 per cent) of compensation area households were recipients and 365 (46 per cent) were non-recipients.

Within the selected districts, sub-districts (*upazila*) with a high concentration of fishers were identified through consultations with District Fisheries Officers (DFOs) and *Upazila* Fisheries Officers (UFOs). Due to difficulties achieving a random and conventional sample design, we aimed instead to interview a large enough proportion of the total fishers in each *upazila*, so that any problems associated with small and non-random sampling were not likely to influence results. For each selected *upazila*, villages with a high concentration of hilsa fishers and preferably located near the river were then selected, based on the list of fishers. Each *upazila* sample was then allocated proportionately to the selected villages, according to the total number of fishers in the village. All households in the selected village were serially numbered and every n^{th} household was selected for interview, where n is the total number of fishers in the village divided by the total *upazila* sample size[39]. Within a household, enumerators interviewed the fisher, if available. If not, they chose a male household member over a female member, since they tend to be more involved in fishing activities. If the enumerators could not find anyone to speak to in the selected household, or if the ratio of recipients to non-recipients in

[39] For example, in Chandpur District, two villages were selected; Lalpur and Gobindia villages. The total number of hilsa fishers in Lalpur and Gobindia were 143 and 737 respectively. The total number of hilsa fishers (880) was therefore divided by 150 (target sample size for Chandpur district). Therefore, an interval of 6 was used to select interviewees from the list of fishers.

compensation areas was becoming unbalanced, they selected the preceding or following fisher on the list.

Estimates of the total number of hilsa fishers in Bangladesh range from 300,000 (A. Wahab 2015, WorldFish, personal communication, 20th March) to 500,000 (M. Mome 2015, Department of Fisheries, personal communication, 20th March) and according to M. Mome, 224,102 households received compensation in 2014 – around 45-75 per cent of affected households, or 65 per cent according to Rahman et al. (2014). 58 per cent of households received compensation in the study sites (Table B.1), and 54 per cent of the households surveyed received compensation, indicating that the sample is roughly representative of the recipient and non-recipient groups.

Table B.1: Details of survey sites with the total numbers of fishing households in each, the numbers of compensation recipients and non-recipients, and sample sizes. Village statistics were obtained from DoF (2014).

District	*Upazila*	Village	Fishing households	Recipients	Non- recipients	Sample size		
						Recipients	Non-recipients	Total
Chandpur	Chandpur Sadar	Lalpur	143	143	0	25	0	25
		Uttar Gabindia	737	737	0	125	0	125
		Sub-total	**880**	**880**	**0**	**150**	**0**	**150**
Laxmipur	Ramgati	Sabujgram	1186	782	404	25	48	73
		Char Laxmi	1480	1190	290	25	52	77
		Sub-total	**2666**	**1972**	**694**	**50**	**100**	**150**
Bhola	Bhola Sadar	Dakhin Razapur	790	425	365	10	30	40
		Kalupur	659	24	635	20	15	35
	Lalmohan	Char Kachopia	865	459	406	16	36	52
		Rayrabad	400	268	132	5	18	23
		Sub-total	**2714**	**1176**	**1538**	**51**	**99**	**150**
Patuakhali	Kalapara	Charipara	320	160	160	5	13	18
		Nizampur	1475	737	738	25	50	75
		Golbunia	302	150	152	5	10	15
		Mewrapara	598	298	300	12	18	30
		Chinguria	280	140	140	5	7	12
		Sub-total	**2975**	**1485**	**1490**	**52**	**98**	**150**
Barisal	Hizla	Baushia	899	727	172	37	20	57
		Hizla Gourbdi	325	263	62	13	5	18
	Muladi	Kutubpur	55	20	35	5	15	20
		Goswherchar	78	40	38	28	2	30
		Sub-total	**133**	**60**	**73**	**83**	**42**	**125**
Barguna	Patharghata	Padma	444	186	258	20	15	35
		Gunpara	515	216	299	29	11	40
		Sub-total	**959**	**402**	**557**	**49**	**26**	**75**
Cox's Bazar	Cox's Bazar	Kutubdiapara	253	0	253	0	26	40
		Nunia Chara	66	0	66	0	24	10
		Sub-total	**319**	**0**	**319**	**0**	**50**	**50**
Rajbari	Gaolondo	Shajapur	92	0	92	0	40	26
		Sorab Mondol Para	84	0	84	0	10	24
		Sub-total	**176**	**0**	**176**	**0**	**50**	**50**
		Grand total	**12046**	**6965**	**5081**	**444**	**456**	**900**

Ethics statement

I followed the ethical principles for research outlined by my collaborators at the International Institute for Environment and Development, which are based on the key principles of respect, beneficence and justice (IIED 2014). All research assistants and participants were informed about the purposes, methods and intended use of research, participation was voluntary, and it was made clear that research was not linked to any immediate benefit for participants. Some KIIs were recorded, with the interviewees consent. Anonymity of participants and confidentiality of information provided were protected, and efforts were made not to inordinately burden participants' time.

Questionnaire

The questionnaire was developed on the basis of focus group discussions previously carried out in five sites from October – December 2013 (BCAS 2013). It was divided into the following 7 sections: household characteristics, fishing activities, hilsa trends and status, the compensation scheme, coping strategy, and opportunity cost questions. A pilot survey was conducted with 28 households in April 2014, feedback and observations from which allowed the questionnaire to be refined. Efforts were taken to identify and minimise sources of bias (Choi & Pak 2005). The questionnaire was finalised in English (below) and translated into Bengali.

DARWIN Initiative: Economic Incentives to Conserve Hilsa Fish (*Tenualosa Ilisha*) in Bangladesh
International Institute for Environment and Development (IIED),
Bangladesh Centre for Advanced Studies (BCAS) and Bangladesh Agricultural University (BAU)

Questionnaire for the socio-economic survey of fishing households

Name of enumerator: ________________ Date: ____________

Code: [number-dd-mm-yy] __________________

District:

Union:

Upazila:

Village:

Village code: Lalpur 1, Uttar Gobindia 2, Sabujgram 3, Char Laxmi 4, Dakhin Razapur 5, Kalupur 6, Charipara 7, Nizampur 8, Baushia 9, Hizla Gourabdi 10, Baherchar 11, Kutubpur 12, Gosherchar 13, Sajapur 14, Sorab Mondal Para 15, Padma 16, Gunpara 17, Kewrabunia 18,Shoto labongola 19, Charkachopia 20, Rayrabad 21, Mewrapara 22, Chinguria 23, Golbunia 24, Other 25 __________

Union Code: Bishnopur 1, Hanerchar 2, Alexander 3, Char Laxmi 4, Dakhin Razapur 5, Ilisha 6, Lalua 7, Mahipur 8, Barojalia 9, Hizla Gourobdi 10, Debogram 11, Doulatdia 12, Muladi sadar 13, Nazirpur 14, Lalmahan 15, Pathorgata sadar 16, Charduani 17, Ailapathagata 18, Burirchar 19, Badarpur 20, Other 21__________

Upazila Code: Chandpur Sadar 1, Ramgati 2, Bhola sadar 3, Doulatkhan 4, Kalapara 5, Hizl -6, Pathargata 7, Cox's Bazar 8, Muladi 9, Rajbari 10, Barguna Sadar 11, Goalondo 12, Lalmahon 13, Other 14__________

District Code: Chandpur 1, Laxmipur 2, Bhola 3, Patualkhali 4, Barisal 5, Barguna 6, Cox's Bazar 7, Rajbari 8, Other 8 __________

Household Characteristics

1. a) Name of household head__________________
   b) Cell number__________________
   c) Name of respondent__________________
   d) Relation to household head________________

2. Gender (0 = Male, 1= Female) _____

3. Age (years) _________

4. Years of schooling_________

5. Number of people currently living in household_________Adult male_________Adult female_________ Children up to 11_________ Adolescents_________ Male ______Female_________

6. Number of other people supported outside of household_________

7. Number of economically active (earning) household members_________

8. Main income generating activity_________

9. Secondary income generating activities_________

(Professional Code Q8 & Q9: Hilsa/Jatka Fishing 1, Other Fishing 2, Day labor 3, Agricultural Activities 4, Fish Trading 5, Service 6, Business 7, Handicrafts 8, Rickshaw/Van puller 9, Farming (Livestock) 10, Farming (Vegetables) 11, Tailoring 12, Net Repairing /Making 13, Foreign currency 14, Boat man 15, Other (Specify) 16

10. Remittance from family members (1= Yes 0= No) _________
    [If yes] How much per month? _________Taka

11. Do you own livestock (1= Yes 0= No) _________
    [If yes] What and how many?
    Cows_________ Chickens_________ Goats_________ Others (Specify) _________

12. Do you have own land (1= Yes 0= No) _________
    [If yes] How much (local units)? Homestead_________ Agricultural_________ Other_________

13. Have you ever borrowed money from any source (1= Yes 0= No) _________
    [If yes] When was the last time and how much? mm-yy _______________ Amount _________Taka
    [If no] Why?
    ____
    ____

14. [if Yes Q13 above, then] who do you usually borrow money from?
    Microfinance institutions (e.g. Grameen Bank or other local MFIs) _________Taka
    Local money lenders (aratdars and dadondars) _________ Taka
    Relatives and friends (no interest rate) _________ Taka
    Other (please specify) _________ Taka

Fishing activities

15. What is the primary purpose of your involvement in fishing?
    1=Subsistence or consumption
    2=Consumption and selling
    3=To sell in the local market
    4=Labour (employed by others)
    5=Other ______________

16. How do you access fishing rights?

    1=Lease

    2= Share

    3= Labour

    4= Contract

    5= License

    6= Free access

17. What is your average monthly income from fishing? ___________Taka

Month/Season	Monthly income (Taka)
March-April	
May-July	
August-October	
November-January	
February	

18. What are your other sources of income of the household?

Source of income	Yearly income (Tk)
Total	

19. Over the last 5 years has your income from fishing changed?

    1= increased
    2= decreased
    3= stayed the same/stable
    4=Not sure Why? ____

20. Are you a member of a fishers association? (1= Yes 0= No)
    If yes, then why?

    1=Access to information/knowledge

2=Access to micro-credit

3=Access to market

4=Access to fishing rights

5=other (please specify) ______________________

21. Do you own a fishing boat? (1= Yes 0= No)

22. Do you catch hilsa fish? (1= Yes 0= No)
    If not, then why?

    ____

    ____

23. What types of fishing gear do you use?

	Name of Gear	Target Fish species (Use code)	Fishing period (month)	Location
1	Chandi Jal -2 (large mess)			
2	Chandi Jal-4 (small mess)			
3	Gulti Jal			
4	Current Jal			
5	Behendi net			
6	Poa Jal			
7	Chapri Net			
8	Chewa net			
9	Chai			
10	Hooks			
11	Moi			
12	Masheri Jal			
13	Khot Jal			
14	Kachki Jal			
15	Bata/goara Jal			

16	Pona Jal			
17	Chargherajal			
18	Cast net			
19	Khota Jal			
20	Scoop net			
21	Dragnet			
22	Lift net/Khora/Beshal jal			
23	Other (Specify)			

Species code: Hilsa 1, Jatka 2, Poa 3, Chewa 4, pangas 5 boal 6 , air 7, baghair 8, chital 9, foli 10, rita 11, puti 12, tengra 13, Golda Chingri 14, Bagda Chingri 15, Bagda 16, kakila 17, baim 18, bele 19, rui 20, Catla 21, mrigel 22, kalibaus 23, miror carp 24, chanda 25, tepa,potka 26, pabda 27, gutum 28, naftani 29, chela 30, mola 31, tatkini 32, bata 33, chapila/mamoli 34, kuchia 35, bacha 36, Kazali/baspata 37, chaka/gangina 38, dela 39, shilong 40, ghaura 41, peali 42, Bajila/batasi 43, tapasi 44, Kholla 45, Small shrimp 46, Kachki 47, Other 48

Month Code: April 1, May 2, June 3, July 4, August 5, September 6,October 7, November 8, December 9, January 10, February 11, March 12

Location code: Meghna River 1, Tetulia River 2, Andermanik River 3, Agunmukha River 4, Sea 5, Padma River 6, Arial Kha River 7, Bishkhali 8, Payra River 9, Ilisha River 10, Jamuna 11, Other (Specify) 12 ___________

Hilsa trends and status

24. How does hilsa vary with season and from year to year?

	Peak season	Lean season	Year to Year (1=Increase 2=Decrease 3=Same 4=don't know)
a) Average catch per fishing trip(kg)			
b) Fish size (1=Small 2=Medium 3=Large)			
c) Presence of eggs/fries (1=Many 2=few 3=none)			

25. Over the last 5 years, has your hilsa catch…

1= Increased

2 = Decreased

3 = Stable

4 = I don't know

26. Over the last 5 years hilsa abundance has…

    1= Increased

    2= Decreased

    3= Stayed stable

    4= I don't know

27. Are you aware of any hilsa management regimes introduced by the Government?
    Jatka fishing ban period (1= Yes 0= No)

    Hilsa sanctuaries (1= Yes 0= No)
    Conservation of gravid hilsa (1= Yes 0= No)

The compensation scheme

28. [Explain the compensation scheme to respondent]

    a) Are you a recipient of the scheme?

    1=Yes 0= No 2= I don't know

    b) How long have you been participating in the scheme? ______________Years

29. Do you think the compensation scheme has had positive impact on hilsa stock regeneration?

    1=Yes 0= No 2= I don't know

30. Do you think the compensation scheme has had positive impact on hilsa catch levels?

    1=Yes 0= No 2= I don't know

31. Do you think the compensation scheme has had positive impact on the improvement of fisher livelihoods?

    1=Yes 0= No 2= I don't know

32. Over the last 5 years, do you think hilsa stocks have changed as a direct result of the compensation scheme?
    1= Increasing 2=Decreasing 3= No change 4= I don’t know

    [If no change] What is the reason?

    ____

    ____

33. Do you think the distribution of compensation is fair? 1=Yes 0= No
    Why/why not? ……………………………………………………………………………….

34. Which people in the village get compensation?

1= Poorest (needy) households

2= Richest households

3= People who fish most

4= People who are most dependent on fishing for their livelihoods

5= People who are part of fisheries association

6= People who are well connected in the village)

7= I don't know

35. Who do you think should be receiving compensation? (Choose any)

1= Poorest households

2= People who are most dependent on fishing for their livelihoods

3= All fishermen should receive compensation

4= I don't know

36. In your opinion, what is the level of compliance with the ban period and zone among:

Fisher type		Degree of compliance [tick one for each type of fisher]				
		1=Everybody complies	2=Most people comply	3=Some people comply	4=Only few people comply	5=No one complies
a) Subsistence fishermen						
Commercial fishermen	b) Recipient					
	c) Non-recipient					
d) Baddya						

37. If not everyone complies in a particular group, then (in your opinion) what needs to be done to enhance compliance within these groups?

38. Do you think the pressure to repay loan is an obstruction to abide with ban period?

1= Yes 0 = No

39. Do you think the fishing ban period fits well with the hilsa's breeding season?

1= Yes 0 = No 2= I don't know

Coping strategy

40. Is your livelihood affected by the closed/off season and zone?

1=Yes 0 = No

41. How sufficient is the compensation provided?

1= More than enough 2=Just enough 3= Not enough

42. If you are not a recipient or the compensation is not sufficient, what are the most important strategies you do to cope in times of hardship? Please rank the 5 most frequently used coping strategies.

Coping strategies	Ranking
1=Use my savings	
2=Borrow from relatives/friends/others	
3=Loan with interest	
4=Shopping credit	
5=Do some other job (e.g. labour or rickshaw/petty business)	
6=Consume less preferred and less expensive food	
7=Limit portion size at meal times	
8=Sell some assets (livestock if any)	
9=Other (specify)	

Opportunity Cost Questions

43. Has not being allowed to catch hilsa in particular zones during the ban period changed your fishing behaviour?

1 = Yes 0 = No

If, yes, then

1= changed location

2= changed location and gears

3= changed target species

4= don't fish during the ban period

5= I go fishing anyway

6=others/specify) ________________

44. If so, has this had an effect on your household income?

1=Increased 2=decreased 3=stayed the same

4=not sure

45. How much in household income do you lose directly due to the ban period and zone? This includes wages (if employed by other fishermen), or loss in earnings from selling fish.

______________ Taka per closure period [Jatka]

_______________ Taka per closure period [brood]

46. How much in household income do you get from other activities that you do during the ban period, which you wouldn't have done if you had been fishing (not things that your household would have done anyway)?
_______________ Taka per closure period

47. If you are affected by the current ban period (or if you were to be affected in the future) how much is your minimum willingness to accept compensation in Taka?
___________Taka per year

This questionnaire was followed by a choice experiment, which was not used in this PhD.

B.2 Model selection tables and results

Table B.2: Model selection table for Model 1: LMM for ecological impact with sanctuary and compensation recipient as fixed effects.

(Intercept)	Boat ownership	Compensation recipient	Sanctuary	Fishing location	Gear diversity	df	logLik	AICc	delta	weight
0.335826	+	NA	NA	+	NA	6	-312.876	637.878	0	0.422613
0.060388	+	NA	+	+	NA	7	-311.871	637.9109	0.032885	0.415721
0.026273	+	+	+	+	NA	8	-312.878	641.9751	4.097111	0.054484
0.309627	+	+	NA	+	NA	7	-314.04	642.2489	4.370833	0.047515
0.317267	+	NA	NA	+	+	7	-314.617	643.4037	5.525625	0.026673
0.04445	+	NA	+	+	+	8	-313.599	643.417	5.53892	0.026496
0.010026	+	+	+	+	+	9	-314.595	647.4637	9.585711	0.003503
0.291066	+	+	NA	+	+	8	-315.779	647.7767	9.898691	0.002996

Table B.3: Model selection table for Model 2: LMM for ecological impact with compensation area as fixed effects.

(Intercept)	Boat	Compensation area	Fishing location	Gear diversity	df	logLik	AICc	delta	weight
0.342135	+	-	+	NA	7	-336.24	686.6329	0	0.597082
0.535062	+	NA	+	NA	6	-337.8	687.714	1.081091	0.347759
0.325237	+	-	+	+	8	-338.054	692.305	5.672144	0.035022
0.519061	+	NA	+	+	7	-339.629	693.4118	6.778963	0.020137

Table B.4: Model selection table for Model 3: LMM for ecological impact with fishing dependence as a fixed effect.

Intercept	Fishing dependence	df	LogLik	AICc	Delta	Weight
0.6716	NA	4	-366.823	741.7	0.0	0.825
0.6613	0.07323	5	-367.358	744.8	3.1	0.175

Table B.5: Coefficient estimates for **(a)** a binomial generalised linear mixed effects model with the probability of targeting *jatka* as the response variable (0 = *jatka*, 1 = no *jatka*); and **(b)** a linear mixed effects model with reported average catch volume (kg per fishing trip) as the response variable, based on 669 households from 19 villages in 6 districts. Random effects estimates of variance are also presented [with standard deviation]. From these models I extracted and mapped the best linear unbiased predictors, which measured the residual effect associated with the district random effect.

(a) Size selectivity		**(b) Catch volume**		
Fixed effects	***Estimate***	***Fixed effects***	***Estimate***	***t value***
Intercept	-0.016 (0.914)	Intercept	0.809 (0.197)	4.098
Random effects		***Random effects***		
Village	1.539 [1.240]	Village	0.009 [0.095]	
District	5.749 [2.398]	District	0.305 [0.552]	

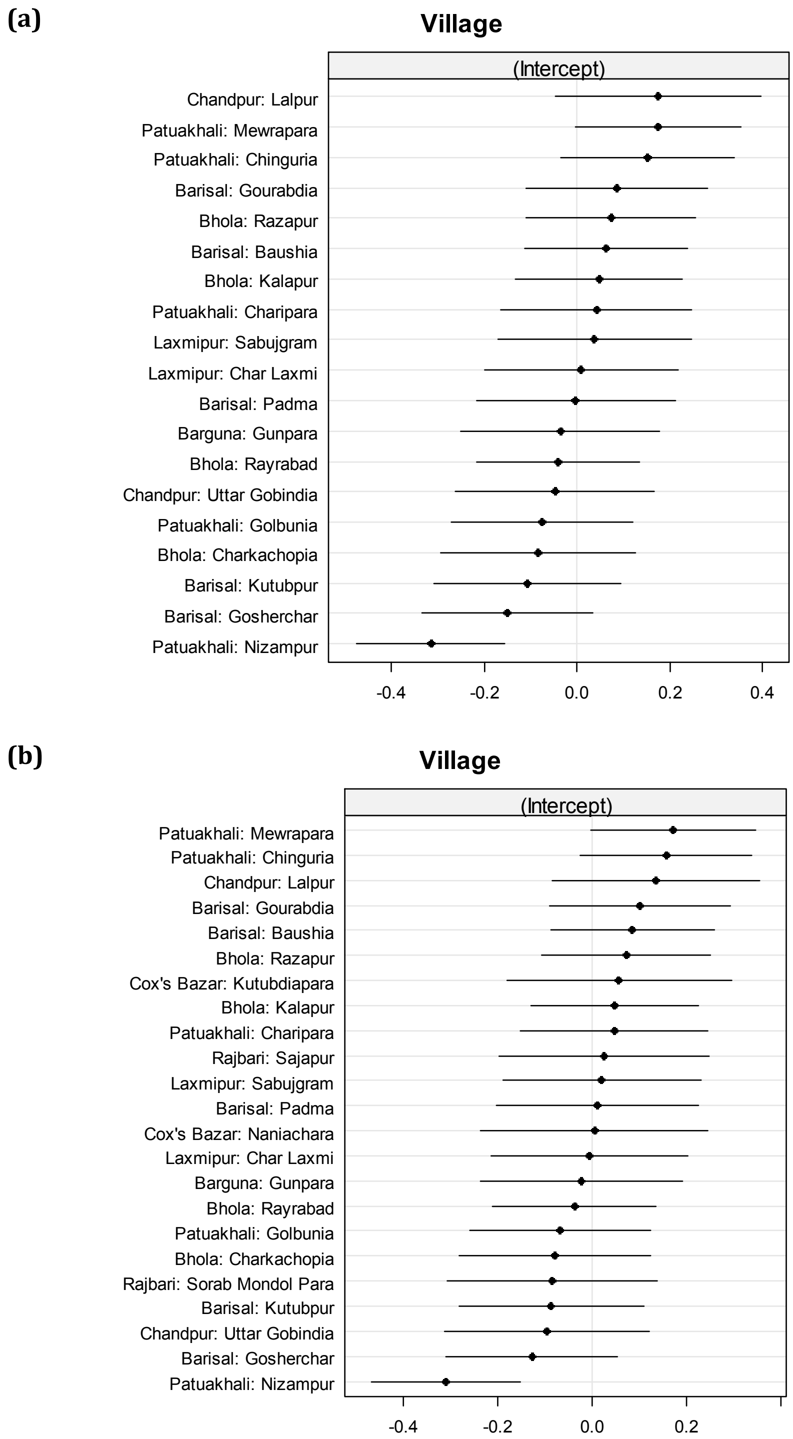

Figure B.3: BLUPs for the random effects of village for **(a)** Model 1 and **(b)** Model 2. The x axes show the effect of living in a particular village in terms of the difference in threat from the intercept. Error bars show the 95% confidence interval based on the conditional variance for each random effect. Village names are prefixed by district.

Appendix B references

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IIED (2014) *Research ethics: putting our principles into practice*. London, International Institute for Environment and Development.

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Appendix C: Supplementary material for Chapter 5

C.1 Key informant interview topics

1. Recent trends in hilsa stock and production
2. Constraints, limitations and strengths of current hilsa fisheries management
3. Opportunities for improved hilsa fisheries management
4. Opinion of the compensation scheme and impact on the hilsa fisheries and livelihoods
5. Enforcement of and compliance with regulations
6. Impact of current management on livelihoods

C.2 Household survey questions used in this chapter (see Appendix B.1 for full questionnaire)

14. Relation to household head________________

15. Gender (0 = Male, 1= Female) _____

16. Age (years) _________

17. Years of schooling_________

15. What is the primary purpose of your involvement in fishing?
    1=Subsistence or consumption
    2=Consumption and selling
    3=To sell in the local market
    4=Labour (employed by others)
    5=Other _____________

19. Over the last 5 years has your income from fishing changed?
    1= increased
    2= decreased
    3= stayed the same/stable
    4=Not sure Why? ____

20. Are you a member of a fishers association? (1= Yes 0= No)
    If yes, then why?
    1=Access to information/knowledge
    2=Access to micro-credit
    3=Access to market
    4=Access to fishing rights
    5=other (please specify) ________________________

21. Do you own a fishing boat? (1= Yes 0= No)

22. Do you catch hilsa fish? (1= Yes 0= No)
    If not, then why?
    ____
    ____

23. What types of fishing gear do you use?

	Name of Gear	Target Fish species (Use code)	Fishing period (month)	Location
1	Chandi Jal -2 (large mess)			
2	Chandi Jal-4 (small mess)			
3	Gulti Jal			
4	Current Jal			
5	Behendi net			
6	Poa Jal			
7	Chapri Net			
8	Chewa net			
9	Chai			
10	Hooks			
11	Moi			
12	Masheri Jal			
13	Khot Jal			
14	Kachki Jal			
15	Bata/goara Jal			
16	Pona Jal			
17	Chargherajal			
18	Cast net			
19	Khota Jal			
20	Scoop net			
21	Dragnet			
22	Lift net/Khora/Beshal jal			

23	Other (Specify)			

24. How does hilsa vary with season and from year to year?

	Peak season	Lean season	Year to Year (1=Increase 2=Decrease 3=Same 4=don't know)
a) Average catch per fishing trip(kg)			
b) Fish size (1=Small 2=Medium 3=Large)			
c) Presence of eggs/fries (1=Many 2=few 3=none)			

25. Over the last 5 years, has your hilsa catch…

    1= Increased

    2 = Decreased

    3 = Stable

    4 = I don't know

26. Over the last 5 years hilsa abundance has…

    1= Increased

    2= Decreased

    3= Stayed stable

    4= I don't know

27. Are you aware of any hilsa management regimes introduced by the Government?
    Jatka fishing ban period (1= Yes 0= No)

    Hilsa sanctuaries (1= Yes 0= No)
    Conservation of gravid hilsa (1= Yes 0= No)

28. [Explain the compensation scheme to respondent]

    a) Are you a recipient of the scheme?

    1=Yes 0= No 2= I don't know

    b) How long have you been participating in the scheme? ______________Years

29. Do you think the compensation scheme has had positive impact on hilsa stock regeneration?

    1=Yes 0= No 2= I don't know

30. Do you think the compensation scheme has had positive impact on hilsa catch levels?

1=Yes 0= No 2= I don't know

31. Do you think the compensation scheme has had positive impact on the improvement of fisher livelihoods?

    1=Yes 0= No 2= I don't know

32. Over the last 5 years, do you think hilsa stocks have changed as a direct result of the compensation scheme?

    1= Increasing 2=Decreasing 3= No change 4= I don't know

    [If no change] What is the reason?

    ____

    ____

36. In your opinion, what is the level of compliance with the ban period and zone among:

Fisher type		Degree of compliance [tick one for each type of fisher]				
		1=Everybody complies	2=Most people comply	3=Some people comply	4=Only few people comply	5=No one complies
Commercial fishermen	b) Recipient					
	c) Non-recipient					

37. If not everyone complies in a particular group, then (in your opinion) what needs to be done to enhance compliance within these groups?

____

____

____

39. Do you think the fishing ban period fits well with the hilsa's breeding season?

    1= Yes 0 = No 2= I don't know

42. If you are not a recipient or the compensation is not sufficient, what are the most important strategies you do to cope in times of hardship? Please rank the 5 most frequently used coping strategies.

Coping strategies	**Ranking**
1=Use my savings	
2=Borrow from relatives/friends/others	
3=Loan with interest	
4=Shopping credit	
5=Do some other job (e.g. labour or rickshaw/petty business)	
6=Consume less preferred and less expensive food	
7=Limit portion size at meal times	
8=Sell some assets (livestock if any)	
9=Other (specify)	

C.3 GLMM results

Table C.1: Results for GLMMs of probability of reporting an increase in **(a)** hilsa catch volume from year to year, where compensation area is included as an explanatory variable instead of sanctuary and compensation; **(b)** hilsa egg/fry presence from year to year, where compensation area is included as an explanatory variable instead of sanctuary and compensation; and **(c)** hilsa egg/fry presence from year to year, where sanctuary and compensation are included as explanatory variables and district random effect is removed. Tables show the full model-averaged coefficient estimates (standard error) and relative importance of each variable from the candidate set of models where **Δ**AICc <4, based on 739 households from 23 villages in 8 districts. Coefficient estimates are presented as contrasts from the intercept, standardised on 2 standard deviations following Gelman (2008). Where the relative importance of a variable is < 0.5, only the direction of the effect is presented. Random effects estimates of variance [standard deviation] were taken from the global model.

(a)

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	2.92 (0.81)	
Age	-0.18 (0.24)	0.50
Compensation area (1 = compensation area, 0 = control)	+	0.38
Awareness (1 = aware of regulations, 0 = not aware)	-	0.33
Target *jatka* (1 = target *jatka*, 0 = no *jatka*)	+	0.29
Fishing location (1 = sea, 0 = only river)	-	0.28
# of models in candidate set	27	
Random effects		
Village	2.52 [1.59]	
District	2.83 [1.68]	

(b)

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	-0.17 (0.19)	
Compensation area (1 = compensation area, 0 = control)	0.83 (0.63)	0.78
Fishing location (1 = sea, 0 = only river)	+	0.36
Age	-	0.30
Target jatka (1 = target jatka, 0 = no jatka)	-	0.24
Awareness (1 = aware of regulations, 0 = not aware)	+	0.19
# of models in candidate set	18	
Random effects		
Village	0.52 [0.72]	
District	0.02 [0.12]	

(c)

Fixed effects	***Estimate (SE)***	***Relative importance***
Intercept	-0.13 (0.17)	
Sanctuary (1 = inside, 0 = outside)	1.08 (0.38)	1.00
Fishing location (1 = sea, 0 = only river)	+	0.38
Awareness (1 = aware of regulations, 0 = not aware)	-	0.34
Age	-	0.27
Target *jatka* (1 = target *jatka*, 0 = no *jatka*)	-	0.21
Compensation (1 = recipient, 0 = non-recipient)	+	0.20
# of models in candidate set	19	
Random effects		
Village	0.44 [0.66]	

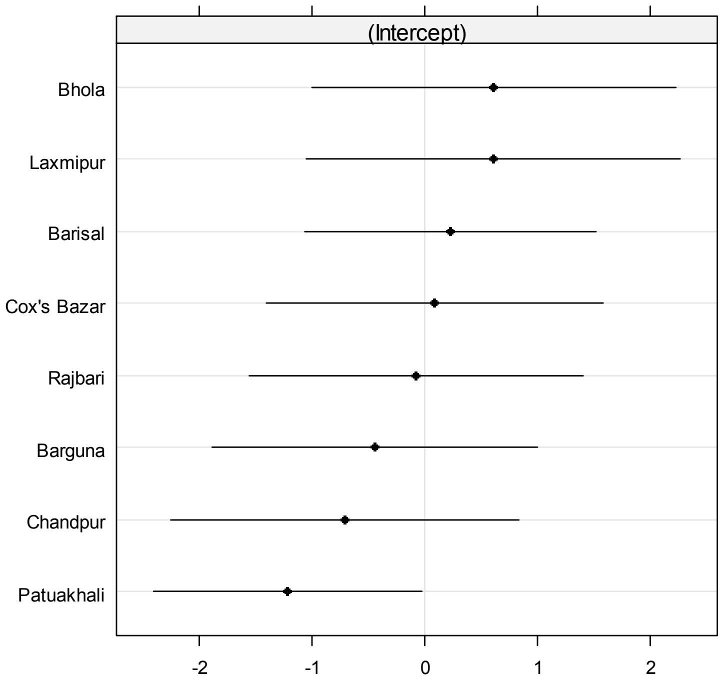

Figure C.1: BLUPs for the district random effect in the GLMM for probability of reporting an increase in catch volume, where sanctuary is included as an explanatory variable. The x axis shows the effect of living in a particular district in terms of the difference in probability of perceiving an increase in catch volume from the intercept. Error bars show the 95% confidence interval based on the conditional variance for the random effect.

C.4 Model selection tables

Table C.2: Model selection table for GLMM with the probability of perceiving an increase in hilsa catch (with compensation area as an explanatory variable).

(Intercept)	Age	Compensation Area	Jatka fishing	Fishing location	Awareness	df	logLik	AICc	delta	weight
2.826258	-0.3394	NA	NA	NA	NA	4	-259.52	527.0948	0	0.09613
2.813248	NA	NA	NA	NA	NA	3	-260.564	527.1603	0.065526	0.093031
3.113442	-0.33935	1.8701	NA	NA	NA	5	-258.96	528.0019	0.907101	0.061078
3.059488	NA	1.81908	NA	NA	NA	4	-260.006	528.0658	0.971083	0.059155
2.815681	-0.34948	NA	-0.35849	NA	NA	5	-259.064	528.2104	1.115607	0.055031
2.802082	NA	NA	-0.33503	NA	NA	4	-260.165	528.3855	1.29071	0.050418
2.814638	NA	NA	NA	0.24045	NA	4	-260.237	528.528	1.433262	0.046949
2.791257	NA	NA	NA	NA	-0.4561	4	-260.244	528.543	1.448265	0.046598
2.805155	-0.32974	NA	NA	NA	-0.41244	5	-259.262	528.606	1.511196	0.045155
2.825568	-0.32885	NA	NA	0.213935	NA	5	-259.264	528.609	1.514218	0.045087
3.117111	-0.34973	2.0066	-0.37504	NA	NA	6	-258.462	529.038	1.943269	0.036382
3.103196	NA	1.98885	-0.35176	NA	NA	5	-259.559	529.2002	2.105489	0.033547
3.057767	NA	1.80615	NA	0.24218	NA	5	-259.674	529.4307	2.335892	0.029897
3.051	NA	1.72182	NA	NA	-0.45365	5	-259.686	529.4532	2.358439	0.029562
3.109375	-0.3286	1.85597	NA	0.215634	NA	6	-258.7	529.5147	2.419991	0.028666
3.070081	-0.32982	1.74637	NA	NA	-0.40994	6	-258.707	529.5295	2.434729	0.028455
2.816938	-0.33948	NA	-0.34513	0.199088	NA	6	-258.843	529.801	2.706259	0.024843
2.80559	NA	NA	-0.31993	0.227009	NA	5	-259.875	529.8323	2.737505	0.024458
2.800501	-0.34099	NA	-0.3249	NA	-0.33539	6	-258.897	529.9092	2.81447	0.023534
2.785837	NA	NA	-0.2959	NA	-0.38594	5	-259.942	529.965	2.870222	0.022887
2.796321	NA	NA	NA	0.216912	-0.40863	5	-259.98	530.0428	2.948037	0.022014
2.807888	-0.32122	NA	NA	0.193456	-0.37195	6	-259.054	530.2225	3.127702	0.020123
3.119486	-0.33956	1.99399	-0.36089	0.199566	NA	7	-258.24	530.6335	3.538726	0.016384
3.108455	NA	1.99744	-0.33687	0.225697	NA	6	-259.269	530.6521	3.557364	0.016233
3.087821	-0.34144	1.90063	-0.34169	NA	-0.32819	7	-258.303	530.7592	3.664476	0.015386

3.066103	NA	1.86416	-0.31328	NA	-0.37863	6	-259.346	530.8059	3.711187	0.015031
3.056712	NA	1.72720	NA	0.218614	-0.40447	6	-259.419	530.9527	3.857962	0.013967

Table C.3: Model selection table for GLMM with the probability of reporting an increase in hilsa catch (with sanctuary and compensation recipient as explanatory variables instead of compensation area)

(Intercept)	Sanctuary	Age	Awareness	Jatka Fishing	Fishing location	Compensation	df	logLik	AICc	delta	weight
3.258244	2.830415	-0.3382	NA	NA	NA	NA	5	-257.204	524.4894	0	0.08635
3.238005	2.798665	NA	NA	NA	NA	NA	4	-258.235	524.5236	0.0342	0.084886
3.24337	3.084928	-0.35004	-0.48111	NA	NA	NA	6	-256.386	524.8863	0.396933	0.070806
3.220821	3.040622	NA	-0.46096	NA	NA	NA	5	-257.484	525.0503	0.560892	0.065233
3.220878	2.790783	NA	NA	0.285709	NA	NA	5	-257.78	525.6409	1.151546	0.048552
3.151212	2.564772	NA	NA	NA	-0.54009	NA	5	-257.807	525.6952	1.205761	0.047254
3.240401	2.820663	-0.32489	NA	0.258141	NA	NA	6	-256.835	525.7857	1.296347	0.045161
3.177693	2.615334	-0.32628	NA	NA	-0.49327	NA	6	-256.851	525.8161	1.326686	0.044481
3.20839	3.030023	NA	-0.44427	0.268354	NA	NA	6	-257.083	526.281	1.791633	0.035255
3.229957	3.073038	-0.33761	-0.46612	0.239641	NA	NA	7	-256.069	526.291	1.801617	0.035079
3.265779	2.819584	-0.3422	NA	NA	NA	0.117023	6	-257.143	526.4009	1.911459	0.033204
3.244529	2.791017	NA	NA	NA	NA	0.093902	5	-258.195	526.4725	1.983109	0.032036
3.182327	2.899275	-0.34004	-0.44532	NA	-0.39535	NA	7	-256.16	526.4739	1.984452	0.032014
3.153133	2.831591	NA	-0.41995	NA	-0.44717	NA	6	-257.192	526.4997	2.010298	0.031603
3.249293	3.073639	-0.35292	-0.47413	NA	NA	0.085298	7	-256.354	526.8606	2.371251	0.026385
3.147322	2.587195	NA	NA	0.259288	-0.48355	NA	6	-257.434	526.983	2.493576	0.024819
3.225505	3.03312	NA	-0.45557	NA	NA	0.061609	6	-257.467	527.0496	2.56023	0.024006
2.826258	NA	-0.3394	NA	NA	NA	NA	4	-259.52	527.0948	2.60536	0.02347
2.813248	NA	NA	NA	NA	NA	NA	3	-260.564	527.1603	2.670885	0.022714
3.171769	2.632145	-0.31539	NA	0.23504	-0.44463	NA	7	-256.547	527.247	2.757581	0.02175
3.227546	2.784874	NA	NA	0.283906	NA	0.08631	6	-257.746	527.6074	3.117971	0.018164
3.158885	2.559061	NA	NA	NA	-0.54054	0.095387	6	-257.766	527.6464	3.156957	0.017813
3.24827	2.811792	-0.32891	NA	0.25578	NA	0.110041	7	-256.782	527.7166	3.227223	0.017198
3.186309	2.60612	-0.33055	NA	NA	-0.49453	0.118822	7	-256.788	527.7284	3.238982	0.017097

Table C.4: Model selection table for GLMM with the probability of reporting an increase in egg/fry presence (with compensation area as an explanatory variable)

(Intercept)	Compensation area	Fishing location	Age	Jatka fishing	Awareness	df	logLik	AICc	delta	weight
-0.15835	1.05392	NA	NA	NA	NA	4	-486.542	981.1381	0	0.168209
-0.15712	1.10616	0.30335	NA	NA	NA	5	-485.9	981.882	0.743901	0.115961
-0.15908	1.05425	NA	-0.13224	NA	NA	5	-486.203	982.4871	1.34903	0.085686
-0.15708	1.05521	NA	NA	-0.11859	NA	5	-486.343	982.768	1.629924	0.074459
-0.15526	1.0111	NA	NA	NA	0.105753	5	-486.474	983.0289	1.890823	0.065352
-0.24305	NA	NA	NA	NA	NA	3	-488.518	983.0682	1.930117	0.064081
-0.15883	1.10655	0.313363	-0.13846	NA	NA	6	-485.532	983.1786	2.040519	0.060639
-0.15667	1.1034	0.280922	NA	-0.07526	NA	6	-485.825	983.7643	2.62616	0.045246
-0.25651	NA	0.321967	NA	NA	NA	4	-487.87	983.7938	2.655669	0.044584
-0.15406	1.08832	0.290882	NA	NA	0.047319	6	-485.89	983.8938	2.755692	0.042409
-0.15777	1.0555	NA	-0.13046	-0.11592	NA	6	-486.013	984.1416	3.003488	0.037467
-0.24493	NA	NA	-0.13722	NA	NA	4	-488.155	984.3642	3.226075	0.033521
-0.15604	1.0122	NA	-0.13153	NA	0.10333	6	-486.138	984.3902	3.252071	0.033088
-0.24229	NA	NA	NA	-0.1175	NA	4	-488.331	984.7163	3.578203	0.028109
-0.15485	1.02381	NA	NA	-0.10997	0.078061	6	-486.307	984.7281	3.590009	0.027944
-0.219	NA	NA	NA	NA	0.14841	4	-488.428	984.9102	3.772058	0.025513
-0.25936	NA	0.334775	-0.14586	NA	NA	5	-487.463	985.0071	3.869021	0.024305

Table C.5: Model selection table for GLMM with the probability of perceiving an increase in egg presence (with sanctuary and compensation recipient as explanatory variables instead of compensation area.)

(Intercept)	Sanctuary	Fishing location	Awareness	Age	Compensation recipient	Jatka Fishing	df	logLik	AICc	delta	weight
-0.13172	1.000132	NA	NA	NA	NA	NA	4	-484.517	977.0891	0	0.157312
-0.12909	1.033748	0.275942	NA	NA	NA	NA	5	-483.946	977.973	0.883867	0.101119
-0.13268	1.187963	NA	-0.27596	NA	NA	NA	5	-484.148	978.3774	1.288296	0.082606
-0.13237	1.00276	NA	NA	-0.13429	NA	NA	5	-484.168	978.4185	1.329346	0.080928
-0.12999	1.278181	0.327213	-0.35406	NA	NA	NA	6	-483.364	978.842	1.752821	0.065485
-0.13179	0.990478	NA	NA	NA	-0.05805	NA	5	-484.469	979.0208	1.931669	0.059883
-0.13076	1.00021	NA	NA	NA	NA	0.015487	5	-484.514	979.1095	2.020353	0.057286
-0.12959	1.037645	0.281196	NA	-0.13801	NA	NA	6	-483.579	979.2726	2.183472	0.052799
-0.13338	1.192036	NA	-0.2782	-0.13533	NA	NA	6	-483.793	979.7011	2.612001	0.042616
-0.12912	1.03144	0.272298	NA	NA	-0.01186	NA	6	-483.944	980.0022	2.913017	0.036661
-0.1289	1.03369	0.275668	NA	NA	NA	0.002986	6	-483.945	980.0056	2.916509	0.036597
-0.13053	1.284117	0.332665	-0.35716	-0.13973	NA	NA	7	-482.988	980.1286	3.03947	0.034415
-0.13292	1.186225	NA	-0.29388	NA	-0.08024	NA	6	-484.058	980.2303	3.141169	0.032709
-0.13243	0.993618	NA	NA	-0.13341	-0.05508	NA	6	-484.125	980.3654	3.276265	0.030572
-0.13248	1.187664	NA	-0.27549	NA	NA	0.003203	6	-484.148	980.41	3.320896	0.029898
-0.13038	1.002973	NA	NA	-0.13728	NA	0.032319	6	-484.153	980.4213	3.332202	0.029729
-0.13013	1.275184	0.318516	-0.3583	NA	-0.02964	NA	7	-483.352	980.8572	3.768073	0.023908
-0.13095	1.280185	0.329027	-0.35663	NA	NA	-0.01564	7	-483.36	980.8734	3.78429	0.023714
-0.13075	0.990469	NA	NA	NA	-0.0585	0.017067	6	-484.465	981.0452	3.956085	0.021762
-0.13172	1.000132	NA	NA	NA	NA	NA	4	-484.517	977.0891	0	0.157312
-0.12909	1.033748	0.275942	NA	NA	NA	NA	5	-483.946	977.973	0.883867	0.101119
-0.13268	1.187963	NA	-0.27596	NA	NA	NA	5	-484.148	978.3774	1.288296	0.082606
-0.13237	1.00276	NA	NA	-0.13429	NA	NA	5	-484.168	978.4185	1.329346	0.080928
-0.12999	1.278181	0.327213	-0.35406	NA	NA	NA	6	-483.364	978.842	1.752821	0.065485

Appendix C references

Gelman, A. (2008) Scaling regression inputs by dividing by two standard deviations. *Statistics in Medicine* 27 (15), 2865–2873. Available from: doi:10.1002/sim.3107 [Accessed: October 1st 2014].

Appendix D: Supplementary material for Chapter 6

D.1 Household survey questions used in this chapter (see Appendix B.1 for full questionnaire)

Household Characteristics

18. Relation to household head________________

19. Gender (0 = Male, 1= Female) _____

20. Age (years) _________

21. Years of schooling_________

22. Number of people currently living in household_________Adult male_________Adult female_________ Children up to 11_________ Adolescents_________ Male ______Female_________

23. Number of other people supported outside of household_________

24. Number of economically active (earning) household members_________

25. Main income generating activity_________

26. Secondary income generating activities_________

27. Remittance from family members (1= Yes 0= No) _________
    [If yes] How much per month? _________Taka

28. Do you own livestock (1= Yes 0= No) _________
    [If yes] What and how many?
    Cows_________ Chickens_________ Goats_________ Others (Specify) _________

29. Do you have own land (1= Yes 0= No) _________
    [If yes] How much (local units)? Homestead_________ Agricultural_________ Other_________

30. Have you ever borrowed money from any source (1= Yes 0= No) _________
    [If yes] When was the last time and how much? mm-yy ______________ Amount _________Taka
    [If no] Why?
    ____
    ____

14. [if Yes Q13 above, then] who do you usually borrow money from?
    Microfinance institutions (e.g. Grameen Bank or other local MFIs) _________Taka

    Local money lenders (aratdars and dadondars) _________ Taka

    Relatives and friends (no interest rate) _________ Taka

    Other (please specify) _________ Taka

Fishing activities

15. What is the primary purpose of your involvement in fishing?
    1=Subsistence or consumption

    2=Consumption and selling

    3=To sell in the local market

    4=Labour (employed by others)
    5=Other ______________

17. What is your average monthly income from fishing? ___________Taka

18. What are your other sources of income of the household?

Source of income	Yearly income (Tk)
Total	

20. Are you a member of a fishers association? (1= Yes 0= No)
    If yes, then why?

    1=Access to information/knowledge

    2=Access to micro-credit

    3=Access to market

    4=Access to fishing rights

    5=other (please specify) ________________________

21. Do you own a fishing boat? (1= Yes 0= No)

22. Do you catch hilsa fish? (1= Yes 0= No)

If not, then why?

____

____

23. What types of fishing gear do you use?

	Name of Gear	Target Fish species (Use code)	Fishing period (month)	Location
1	Chandi Jal -2 (large mess)			
2	Chandi Jal-4 (small mess)			
3	Gulti Jal			
4	Current Jal			
5	Behendi net			
6	Poa Jal			
7	Chapri Net			
8	Chewa net			
9	Chai			
10	Hooks			
11	Moi			
12	Masheri Jal			
13	Khot Jal			
14	Kachki Jal			
15	Bata/goara Jal			
16	Pona Jal			
17	Chargherajal			
18	Cast net			
19	Khota Jal			
20	Scoop net			

21	Dragnet			
22	Lift net/Khora/Beshal jal			
23	Other (Specify)			

The compensation scheme

28. [Explain the compensation scheme to respondent]

a) Are you a recipient of the scheme?

1=Yes 0= No 2= I don't know

b) How long have you been participating in the scheme? ______________Years

33. Do you think the distribution of compensation is fair? 1=Yes 0= No
Why/why not? ……………………………………………………………………………….

34. Which people in the village get compensation?

1= Poorest (needy) households

2= Richest households

3= People who fish most

4= People who are most dependent on fishing for their livelihoods

5= People who are part of fishers associations

6= People who are well connected in the village)

7= I don't know

35. Who do you think should be receiving compensation? (Choose any)

1= Poorest households

2= People who are most dependent on fishing for their livelihoods

3= All fishermen should receive compensation

4= I don't know

Coping strategy

40. Is your livelihood affected by the closed/off season and zone?

1=Yes 0 = No

41. How sufficient is the compensation provided?

1= More than enough 2=Just enough 3= Not enough

42. If you are not a recipient or the compensation is not sufficient, what are the most important strategies you do to cope in times of hardship? Please rank the 5 most frequently used coping strategies.

Coping strategies	**Ranking**
1=Use my savings	
2=Borrow from relatives/friends/others	
3=Loan with interest	
4=Shopping credit	
5=Do some other job (e.g. labour or rickshaw/petty business)	
6=Consume less preferred and less expensive food	
7=Limit portion size at meal times	
8=Sell some assets (livestock if any)	
9=Other (specify)	

Table D.1: Details of survey sites with the total numbers of fishing households in each, the numbers of compensation recipients and non-recipients, and sample sizes. Village statistics were obtained from DoF (2014).

District	***Upazila***	**Village**	**Fishing households**	**Recipients**	**Non- recipients**	**Sample size**		
						Recipients	Non-recipients	Total
Chandpur	Chandpur	Lalpur	143	143	0	25	0	25
	Sadar	Uttar Gabindia	737	737	0	125	0	125
		Sub-total	**880**	**880**	**0**	**150**	**0**	**150**
Laxmipur	Ramgati	Sabujgram	1186	782	404	25	48	73
		Char Laxmi	1480	1190	290	25	52	77
		Sub-total	**2666**	**1972**	**694**	**50**	**100**	**150**
Bhola	Bhola Sadar	Dakhin Razapur	790	425	365	10	30	40
		Kalupur	659	24	635	20	15	35
	Lalmohan	Char Kachopia	865	459	406	16	36	52
		Rayrabad	400	268	132	5	18	23
		Sub-total	**2714**	**1176**	**1538**	**51**	**99**	**150**
Patuakhali	Kalapara	Charipara	320	160	160	5	13	18
		Nizampur	1475	737	738	25	50	75
		Golbunia	302	150	152	5	10	15
		Mewrapara	598	298	300	12	18	30
		Chinguria	280	140	140	5	7	12
		Sub-total	**2975**	**1485**	**1490**	**52**	**98**	**150**
Barisal	Hizla	Baushia	899	727	172	37	20	57
		Hizla Gourbdi	325	263	62	13	5	18
	Muladi	Kutubpur	55	20	35	5	15	20
		Goswherchar	78	40	38	28	2	30
		Sub-total	**1357**	**1050**	**307**	**83**	**42**	**125**
Barguna	Patharghata	Padma	444	186	258	20	15	35
		Gunpara	515	216	299	29	11	40
		Sub-total	**959**	**402**	**557**	**49**	**26**	**75**
		Grand total	**11551**	**6965**	**4586**	**435**	**365**	**800**

D.3 Model selection tables

Table D.2: Model selection table for GLMM with the probability of fishing *jatka.*

(Intercept)	Fishing dependence	Income	Dependency ratio	Household Size	Respondent identity	Sanctuary	Fisher association membership	Food insecurity	Debt	df	logLik	AICc	delta	weight
0.111297	0.830196	NA	NA	NA	NA	NA	NA	NA	NA	4	-376.215	760.4814	0	0.048199
0.114365	0.800711	-0.14707	NA	NA	NA	NA	NA	NA	NA	5	-375.436	760.948	0.466595	0.03817
0.116037	0.798804	-0.17991	-0.15055	NA	NA	NA	NA	NA	NA	6	-374.542	761.1913	0.709885	0.033798
0.112111	0.83363	NA	-0.12059	NA	NA	NA	NA	NA	NA	5	-375.62	761.3158	0.834349	0.031759
0.117768	0.791589	-0.24586	-0.19313	0.157198	NA	NA	NA	NA	NA	7	-373.767	761.6761	1.194612	0.026524
0.110669	0.83331	NA	NA	NA	0.110643	NA	NA	NA	NA	5	-375.899	761.875	1.393603	0.024012
0.066151	0.82826	NA	NA	NA	NA	-0.63444	NA	NA	NA	5	-375.944	761.9648	1.483351	0.022958
0.103241	0.833377	NA	NA	NA	NA	NA	0.261075	NA	NA	5	-375.947	761.9697	1.488292	0.022901
0.104447	0.80076	-0.16284	NA	NA	NA	NA	0.328748	NA	NA	6	-375.019	762.1451	1.663645	0.020979
0.113847	0.802167	-0.15721	NA	NA	0.126837	NA	NA	NA	NA	6	-375.023	762.154	1.672512	0.020886
0.115083	0.796958	-0.17906	NA	0.096631	NA	NA	NA	NA	NA	6	-375.111	762.3294	1.847956	0.019132
0.111232	0.831205	NA	NA	0.043813	NA	NA	NA	NA	NA	5	-376.139	762.3552	1.873706	0.018887
0.106115	0.798756	-0.19587	-0.15061	NA	NA	NA	0.329189	NA	NA	7	-374.125	762.3937	1.912234	0.018527
0.115452	0.800109	-0.19042	-0.15008	NA	0.125652	NA	NA	NA	NA	7	-374.139	762.4203	1.938898	0.018282
0.111665	0.825053	NA	NA	NA	NA	NA	NA	-0.05078	NA	5	-376.183	762.4418	1.960376	0.018086
0.069674	0.799021	-0.14615	NA	NA	NA	-0.62677	NA	NA	NA	6	-375.174	762.4542	1.972762	0.017975
0.111778	0.829058	NA	NA	NA	NA	NA	NA	NA	0.027715	5	-376.206	762.4875	2.006043	0.017678
0.072425	0.797227	-0.17866	-0.14934	NA	NA	-0.61015	NA	NA	NA	7	-374.294	762.7311	2.249654	0.015651
0.111401	0.836557	NA	-0.11858	NA	0.107004	NA	NA	NA	NA	6	-375.325	762.7571	2.275682	0.015448
0.107219	0.791008	-0.26449	-0.19443	0.16191	NA	NA	0.349926	NA	NA	8	-373.3	762.7843	2.302807	0.01524
0.067744	0.831729	NA	-0.11956	NA	NA	-0.62272	NA	NA	NA	6	-375.359	762.8243	2.342875	0.014938
0.104279	0.836715	NA	-0.11886	NA	NA	NA	0.252563	NA	NA	6	-375.369	762.8443	2.362894	0.014789
0.114831	0.793739	-0.14927	NA	NA	NA	NA	NA	-0.06378	NA	6	-375.385	762.8771	2.395682	0.014549
0.117062	0.792789	-0.25786	-0.19294	0.158567	0.127928	NA	NA	NA	NA	8	-373.349	762.8826	2.401146	0.014509

0.115394	0.797738	-0.15032	NA	NA	NA	NA	NA	NA	0.05666	6	-375.4	762.9076	2.426121	0.014329
0.112036	0.835655	NA	-0.13521	0.073048	NA	NA	NA	NA	NA	6	-375.421	762.9481	2.466682	0.014041
0.117652	0.794118	-0.18623	-0.15495	NA	NA	NA	NA	NA	0.0889	7	-374.461	763.064	2.582523	0.013251
0.116496	0.791871	-0.18227	-0.15044	NA	NA	NA	NA	-0.06273	NA	7	-374.493	763.1284	2.647001	0.012831
0.073074	0.789815	-0.24526	-0.19236	0.159112	NA	-0.62954	NA	NA	NA	8	-373.5	763.1833	2.701838	0.012484
0.112441	0.828799	NA	-0.12019	NA	NA	NA	NA	-0.04741	NA	6	-375.591	763.2892	2.807799	0.011839
0.112897	0.831733	NA	-0.12243	NA	NA	NA	NA	NA	0.048084	6	-375.594	763.2945	2.813018	0.011809
0.065823	0.831359	NA	NA	NA	0.109094	-0.62829	NA	NA	NA	6	-375.637	763.3817	2.900283	0.011304
0.104261	0.802102	-0.17229	NA	NA	0.123136	NA	0.319259	NA	NA	7	-374.631	763.4051	2.923654	0.011173
0.102949	0.836247	NA	NA	NA	0.106631	NA	0.249828	NA	NA	6	-375.654	763.4151	2.933656	0.011117
0.104784	0.79664	-0.19701	NA	0.101178	NA	NA	0.341975	NA	NA	7	-374.663	763.468	2.986506	0.010827
0.057728	0.831445	NA	NA	NA	NA	-0.63637	0.259011	NA	NA	6	-375.681	763.4681	2.986651	0.010827
0.114526	0.798349	-0.19028	NA	0.098392	0.128716	NA	NA	NA	NA	7	-374.687	763.5169	3.035478	0.010566
0.119417	0.787118	-0.25228	-0.19735	0.156668	NA	NA	NA	NA	0.085626	8	-373.691	763.5659	3.084459	0.01031
0.118225	0.785656	-0.24759	-0.19278	0.156068	NA	NA	NA	-0.05418	NA	8	-373.73	763.6443	3.162865	0.009914
0.059379	0.799132	-0.16174	NA	NA	NA	-0.62825	0.326324	NA	NA	7	-374.764	763.6709	3.189455	0.009783
0.105812	0.799965	-0.20569	-0.15021	NA	0.122079	NA	0.320126	NA	NA	8	-373.745	763.6747	3.193264	0.009764
0.069531	0.80048	-0.15621	NA	NA	0.125284	-0.61948	NA	NA	NA	7	-374.772	763.6862	3.204744	0.009708
0.110548	0.834284	NA	NA	0.04264	0.110028	NA	NA	NA	NA	6	-375.827	763.7615	3.280053	0.009349
0.069493	0.795091	-0.17882	NA	0.098868	NA	-0.64195	NA	NA	NA	7	-374.834	763.8111	3.329642	0.00912
0.065593	0.829272	NA	NA	0.046084	NA	-0.64227	NA	NA	NA	6	-375.86	763.828	3.346536	0.009044
0.103123	0.834383	NA	NA	0.04384	NA	NA	0.261396	NA	NA	6	-375.87	763.8478	3.366385	0.008954
0.110898	0.829888	NA	NA	NA	0.108054	NA	NA	-0.03294	NA	6	-375.885	763.878	3.396548	0.00882
0.111167	0.832068	NA	NA	NA	0.110963	NA	NA	NA	0.030549	6	-375.888	763.8825	3.401055	0.008801
0.103566	0.827982	NA	NA	NA	NA	NA	0.262563	-0.05339	NA	6	-375.911	763.9284	3.446997	0.008601
0.066644	0.823084	NA	NA	NA	NA	-0.63316	NA	-0.05119	NA	6	-375.911	763.9292	3.447731	0.008598
0.06215	0.79723	-0.19445	-0.14942	NA	NA	-0.61142	0.32688	NA	NA	8	-373.884	763.9526	3.471148	0.008497
0.103796	0.831746	NA	NA	NA	NA	NA	0.266643	NA	0.042076	6	-375.926	763.9591	3.47761	0.00847
0.06653	0.827101	NA	NA	NA	NA	-0.63592	NA	NA	0.028518	6	-375.934	763.9749	3.493464	0.008403

0.072264	0.798538	-0.18909	-0.1489	NA	0.124163	-0.60292	NA	NA	NA	8	-373.901	763.9851	3.503651	0.00836
0.106847	0.792121	-0.27555	-0.19422	0.163006	0.124146	NA	0.340524	NA	NA	9	-372.908	764.047	3.565569	0.008106
0.105524	0.796659	-0.16806	NA	NA	NA	NA	0.341358	NA	0.078527	7	-374.953	764.0496	3.568128	0.008095
0.104878	0.793222	-0.16545	NA	NA	NA	NA	0.33172	-0.06864	NA	7	-374.96	764.0634	3.581961	0.008039
0.114965	0.798908	-0.16092	NA	NA	0.127884	NA	NA	NA	0.062188	7	-374.981	764.1052	3.62372	0.007873
0.114178	0.797295	-0.15848	NA	NA	0.123464	NA	NA	-0.04405	NA	7	-374.999	764.1411	3.65969	0.007733
0.107633	0.792907	-0.20481	-0.15615	NA	NA	NA	0.347113	NA	0.111603	8	-374	764.1835	3.702006	0.007571
0.117172	0.795095	-0.19745	-0.15489	NA	0.127398	NA	NA	NA	0.095143	8	-374.046	764.2762	3.794783	0.007228
0.11555	0.790645	-0.18074	NA	0.095426	NA	NA	NA	-0.05833	NA	7	-375.069	764.2811	3.7997	0.00721
0.067341	0.834638	NA	-0.11761	NA	0.105523	-0.61696	NA	NA	NA	7	-375.072	764.2876	3.806191	0.007187
0.06204	0.789304	-0.26365	-0.19365	0.163729	NA	-0.63175	0.347687	NA	NA	9	-373.041	764.3113	3.829878	0.007102
0.115963	0.794447	-0.18158	NA	0.095485	NA	NA	NA	NA	0.048755	7	-375.084	764.3118	3.830313	0.007101
0.106514	0.791252	-0.19864	-0.15049	NA	NA	NA	0.332118	-0.06757	NA	8	-374.068	764.3203	3.838848	0.00707
0.111551	0.826412	NA	NA	0.042489	NA	NA	NA	-0.04713	NA	6	-376.112	764.3301	3.848656	0.007036
0.103967	0.839423	NA	-0.11703	NA	0.103229	NA	0.242015	NA	NA	7	-375.095	764.333	3.851597	0.007025
0.059584	0.834823	NA	-0.11786	NA	NA	-0.62469	0.250672	NA	NA	7	-375.112	764.3669	3.885489	0.006907
0.111595	0.830308	NA	NA	0.043007	NA	NA	NA	NA	0.021449	6	-376.133	764.3735	3.892093	0.006885
0.070308	0.792022	-0.14835	NA	NA	NA	-0.6251	NA	-0.0641	NA	7	-375.122	764.3873	3.905885	0.006837
0.115794	0.795273	-0.19178	-0.15002	NA	0.12236	NA	NA	-0.04328	NA	8	-374.115	764.4141	3.932618	0.006747
0.111308	0.838504	NA	-0.13276	0.071112	0.105297	NA	NA	NA	NA	7	-375.136	764.4144	3.932964	0.006745
0.072764	0.791026	-0.25714	-0.19219	0.160376	0.126388	-0.6219	NA	NA	NA	9	-373.093	764.4164	3.934985	0.006739
0.070486	0.79601	-0.14945	NA	NA	NA	-0.62959	NA	NA	0.057417	7	-375.137	764.4171	3.935612	0.006736
0.06696	0.833728	NA	-0.13458	0.075124	NA	-0.63407	NA	NA	NA	7	-375.148	764.4392	3.957708	0.006662
0.112028	0.824342	NA	NA	NA	NA	NA	NA	-0.0488	0.022158	6	-376.176	764.4595	3.978068	0.006595

Table D.3: Model selection table for GLMM with probability of receiving compensation

(Intercept)	Fisher association membership	Food Insecurity	Dependency ratio	Household Size	Income	Respondent identity	Sanctuary	Jatka fishing	Fishing dependence	Debt	df	logLik	AICc	delta	weight
0.38789	-0.48528	0.264463	-0.17669	0.29694	NA	NA	NA	NA	NA	NA	7	-403.102	820.3462	0	0.030774
0.388714	-0.4754	NA	-0.16933	0.281568	NA	NA	NA	NA	NA	NA	6	-404.208	820.5226	0.176382	0.028176
0.387466	-0.46356	NA	NA	0.239418	NA	NA	NA	NA	NA	NA	5	-405.302	820.6804	0.334117	0.026039
0.386382	-0.47252	0.252911	NA	0.252417	NA	NA	NA	NA	NA	NA	6	-404.289	820.684	0.337801	0.025991
0.389055	-0.53937	0.255025	NA	0.207568	0.186455	NA	NA	NA	NA	NA	7	-403.42	820.9828	0.636584	0.022385
0.390088	-0.52902	NA	NA	0.194767	0.184156	NA	NA	NA	NA	NA	6	-404.449	821.0054	0.659205	0.022133
0.383961	NA	0.257772	-0.17083	0.288434	NA	NA	NA	NA	NA	NA	6	-404.531	821.1689	0.822684	0.020396
0.384843	NA	NA	-0.16377	0.273609	NA	NA	NA	NA	NA	NA	5	-405.582	821.2405	0.894298	0.019678
0.383709	NA	NA	NA	0.232789	NA	NA	NA	NA	NA	NA	4	-406.608	821.2676	0.921325	0.019414
0.382571	NA	0.246746	NA	0.245335	NA	NA	NA	NA	NA	NA	5	-405.643	821.3633	1.017099	0.018506
0.389623	-0.53028	0.264024	-0.14642	0.256244	0.137058	NA	NA	NA	NA	NA	8	-402.668	821.5194	1.173172	0.017117
0.387556	-0.53919	NA	NA	NA	0.24989	NA	NA	NA	NA	NA	5	-405.771	821.6186	1.272394	0.016289
0.390487	-0.52023	NA	-0.13894	0.240601	0.137352	NA	NA	NA	NA	NA	7	-403.769	821.6811	1.334882	0.015788
0.386736	-0.54937	0.23183	NA	NA	0.25581	NA	NA	NA	NA	NA	6	-404.908	821.9222	1.575988	0.013995
0.389379	-0.49125	0.274844	-0.17772	0.2984	NA	0.077311	NA	NA	NA	NA	8	-402.946	822.0763	1.73008	0.012957
0.385056	NA	NA	NA	0.199345	0.136348	NA	NA	NA	NA	NA	5	-406.104	822.2844	1.938169	0.011677
0.370026	-0.48533	0.264904	-0.17656	0.297223	NA	NA	-0.16526	NA	NA	NA	8	-403.091	822.3666	2.020401	0.011206
0.387903	-0.48575	0.264321	-0.17591	0.296071	NA	NA	NA	0.013859	NA	NA	8	-403.097	822.3778	2.031567	0.011144
0.387493	-0.48638	0.262571	-0.17608	0.296617	NA	NA	NA	NA	-0.0118	NA	8	-403.097	822.3779	2.031696	0.011143
0.383928	NA	0.247282	NA	0.211894	0.13716	NA	NA	NA	NA	NA	6	-405.136	822.3783	2.032055	0.011141
0.387933	-0.48509	0.265133	-0.17685	0.29662	NA	NA	NA	NA	NA	0.006047	8	-403.101	822.3865	2.040238	0.011096
0.389737	-0.47931	NA	-0.16996	0.282263	NA	0.054855	NA	NA	NA	NA	7	-404.129	822.4002	2.054	0.011019
0.387677	-0.47827	0.262957	NA	0.253538	NA	0.07354	NA	NA	NA	NA	7	-404.147	822.4372	2.090979	0.010818
0.387441	-0.47859	NA	-0.16777	0.280925	NA	NA	NA	NA	-0.03254	NA	7	-404.173	822.4882	2.141956	0.010545
0.388495	-0.4771	NA	-0.16839	0.284208	NA	NA	NA	NA	NA	-0.0443	7	-404.191	822.5239	2.177708	0.010359
0.373529	-0.47542	NA	-0.16921	0.281791	NA	NA	-0.14105	NA	NA	NA	7	-404.201	822.5441	2.197866	0.010255

0.388784	-0.47595	NA	-0.16844	0.280594	NA	NA	NA	0.015833	NA	NA	7	-404.202	822.546	2.199795	0.010245
0.388388	-0.46732	NA	NA	0.239902	NA	0.051782	NA	NA	NA	NA	6	-405.231	822.5693	2.223062	0.010126
0.385885	-0.46791	NA	NA	0.239211	NA	NA	NA	NA	-0.04209	NA	6	-405.243	822.5938	2.247578	0.010003
0.38716	-0.46599	NA	NA	0.24326	NA	NA	NA	NA	NA	-0.05949	6	-405.271	822.6483	2.302021	0.009734
0.387644	-0.46467	NA	NA	0.237975	NA	NA	NA	0.030435	NA	NA	6	-405.279	822.6649	2.318697	0.009653
0.386404	-0.47359	0.252689	NA	0.251021	NA	NA	NA	0.029167	NA	NA	7	-404.267	822.6777	2.331489	0.009592
0.385561	-0.47475	0.249303	NA	0.252164	NA	NA	NA	NA	-0.02282	NA	7	-404.271	822.6857	2.339427	0.009554
0.371022	-0.46359	NA	NA	0.23969	NA	NA	-0.15265	NA	NA	NA	6	-405.294	822.6941	2.347905	0.009513
0.367282	-0.4726	0.253392	NA	0.252761	NA	NA	-0.17624	NA	NA	NA	7	-404.277	822.6965	2.350277	0.009502
0.386318	-0.47298	0.251568	NA	0.253161	NA	NA	NA	NA	NA	-0.01257	7	-404.287	822.7175	2.371284	0.009403
0.384698	NA	0.256844	-0.15089	0.261582	0.089394	NA	NA	NA	NA	NA	7	-404.33	822.8037	2.457447	0.009006
0.390158	-0.54276	0.263496	NA	0.20956	0.182509	0.062949	NA	NA	NA	NA	8	-403.317	822.8182	2.471942	0.008941
0.385635	NA	NA	-0.14346	0.246199	0.090977	NA	NA	NA	NA	NA	6	-405.373	822.8538	2.507518	0.008784
0.389392	-0.54343	0.254762	NA	0.203244	0.193742	NA	NA	0.053787	NA	NA	8	-403.35	822.8834	2.537134	0.008655
0.390691	-0.53315	NA	NA	0.190418	0.191471	NA	NA	0.054576	NA	NA	7	-404.377	822.896	2.549775	0.0086
0.390845	-0.53102	NA	NA	0.195839	0.181569	0.041315	NA	NA	NA	NA	7	-404.404	822.9517	2.605506	0.008364
0.389753	-0.53217	NA	NA	0.198911	0.185499	NA	NA	NA	NA	-0.06944	7	-404.406	822.9555	2.609248	0.008348
0.385217	NA	0.266915	-0.17172	0.28963	NA	0.06845	NA	NA	NA	NA	7	-404.408	822.9597	2.613443	0.008331
0.389963	-0.53937	0.258764	NA	0.206056	0.193783	NA	NA	NA	0.022945	NA	8	-403.405	822.9933	2.647017	0.008192
0.369474	-0.53947	0.255523	NA	0.207902	0.186538	NA	-0.18068	NA	NA	NA	8	-403.407	822.9989	2.65262	0.008169
0.388922	-0.54028	0.25262	NA	0.208782	0.186868	NA	NA	NA	NA	-0.0224	8	-403.416	823.0157	2.669437	0.008101
0.383205	-0.44016	NA	NA	NA	NA	NA	NA	NA	NA	NA	4	-407.483	823.0176	2.671408	0.008093
0.37326	-0.52909	NA	NA	0.195036	0.184217	NA	-0.15648	NA	NA	NA	7	-404.44	823.0232	2.676986	0.00807
0.390112	-0.52903	NA	NA	0.194725	0.184372	NA	NA	NA	0.00069	NA	7	-404.449	823.0413	2.69506	0.007998
0.382335	NA	NA	NA	NA	0.199784	NA	NA	NA	NA	NA	4	-407.496	823.0438	2.697582	0.007987
0.385722	NA	NA	-0.16429	0.274143	NA	0.046871	NA	NA	NA	NA	6	-405.524	823.1551	2.808852	0.007555
0.383706	NA	0.255559	NA	0.246233	NA	0.064905	NA	NA	NA	NA	6	-405.533	823.1727	2.826436	0.007489
0.365906	NA	0.258209	-0.17071	0.288734	NA	NA	-0.1668	NA	NA	NA	7	-404.521	823.1842	2.837921	0.007446
0.384488	NA	NA	NA	0.233137	NA	0.043994	NA	NA	NA	NA	5	-406.557	823.1903	2.844051	0.007423

0.384092	NA	0.259821	-0.1713	0.287475	NA	NA	NA	NA	NA	0.018537	7	-404.528	823.198	2.851752	0.007395
0.383955	NA	0.25766	-0.17028	0.287839	NA	NA	NA	0.009509	NA	NA	7	-404.529	823.2003	2.854026	0.007386
0.383833	NA	0.257212	-0.17065	0.288328	NA	NA	NA	NA	-0.00339	NA	7	-404.531	823.2039	2.857675	0.007373
0.382418	NA	NA	NA	0.23252	NA	NA	NA	NA	-0.03315	NA	5	-406.572	823.2195	2.873259	0.007316
0.383875	NA	NA	-0.16261	0.273061	NA	NA	NA	NA	-0.02384	NA	6	-405.563	823.2329	2.886673	0.007267
0.384667	NA	NA	-0.1631	0.275376	NA	NA	NA	NA	NA	-0.0307	6	-405.574	823.2547	2.908491	0.007188
0.383424	NA	NA	NA	0.235651	NA	NA	NA	NA	NA	-0.04538	5	-406.59	823.2564	2.910194	0.007182
0.36941	NA	NA	-0.16365	0.273836	NA	NA	-0.14328	NA	NA	NA	6	-405.575	823.2566	2.910379	0.007181
0.383821	NA	NA	NA	0.231558	NA	NA	NA	0.025887	NA	NA	5	-406.592	823.2594	2.913174	0.007171
0.384878	NA	NA	-0.16311	0.272899	NA	NA	NA	0.011475	NA	NA	6	-405.579	823.2646	2.918364	0.007153
0.367107	NA	NA	NA	0.233064	NA	NA	-0.1541	NA	NA	NA	5	-406.6	823.2762	2.929915	0.007111
0.387215	-0.53542	NA	-0.06881	NA	0.234241	NA	NA	NA	NA	NA	6	-405.587	823.2808	2.934549	0.007095
0.390915	-0.53381	0.273242	-0.14838	0.259093	0.13201	0.068831	NA	NA	NA	NA	9	-402.545	823.3208	2.974557	0.006954
0.382585	NA	0.246555	NA	0.244153	NA	NA	NA	0.024619	NA	NA	6	-405.628	823.3637	3.017458	0.006807
0.388643	-0.54492	NA	NA	NA	0.258524	NA	NA	0.076017	NA	NA	6	-405.628	823.364	3.017735	0.006806
0.3634	NA	0.247219	NA	0.245679	NA	NA	-0.177	NA	NA	NA	6	-405.632	823.3709	3.024655	0.006782
0.379845	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	3	-408.673	823.3766	3.030384	0.006763
0.382061	NA	0.24448	NA	0.245132	NA	NA	NA	NA	-0.01411	NA	6	-405.637	823.3806	3.034364	0.00675
0.382585	NA	0.246811	NA	0.245304	NA	NA	NA	NA	NA	0.000567	6	-405.643	823.394	3.047771	0.006704
0.382405	-0.44694	0.221894	NA	NA	NA	NA	NA	NA	NA	NA	5	-406.688	823.453	3.106775	0.00651
0.381456	NA	0.223271	NA	NA	0.203952	NA	NA	NA	NA	NA	5	-406.695	823.4655	3.119291	0.006469
0.389778	-0.5331	0.26365	-0.14324	0.252416	0.142716	NA	NA	0.034669	NA	NA	9	-402.639	823.5079	3.161693	0.006333
0.371155	-0.53038	0.2645	-0.14626	0.256518	0.137194	NA	-0.17053	NA	NA	NA	9	-402.657	823.5434	3.197206	0.006222
0.390382	-0.53021	0.267166	-0.14605	0.254934	0.143253	NA	NA	NA	0.019544	NA	9	-402.657	823.5435	3.197267	0.006222
0.386475	-0.54544	0.233726	-0.07162	NA	0.239451	NA	NA	NA	NA	NA	7	-404.708	823.5598	3.213598	0.006171
0.389586	-0.53046	0.263563	-0.14628	0.256432	0.137188	NA	NA	NA	NA	-0.00427	9	-402.668	823.5655	3.219296	0.006153
0.38821	-0.54075	NA	NA	NA	0.247949	0.03425	NA	NA	NA	NA	6	-405.74	823.5875	3.241224	0.006086
0.391335	-0.52239	NA	-0.14017	0.242228	0.133984	0.046409	NA	NA	NA	NA	8	-403.713	823.6097	3.263437	0.006019
0.388266	-0.53915	NA	NA	NA	0.254945	NA	NA	NA	0.01672	NA	6	-405.763	823.6326	3.286391	0.00595

0.373788	-0.53924	NA	NA	NA	0.25001	NA	-0.12772	NA	NA	NA	6	-405.765	823.6377	3.291476	0.005935
0.387319	-0.54031	NA	NA	NA	0.250875	NA	NA	NA	NA	-0.02493	6	-405.766	823.6384	3.292209	0.005933
0.390807	-0.52319	NA	-0.13562	0.236613	0.143247	NA	NA	0.036578	NA	NA	8	-403.737	823.6576	3.311367	0.005877
0.387554	-0.5552	0.232167	NA	NA	0.264662	NA	NA	0.076662	NA	NA	7	-404.763	823.6688	3.322538	0.005844
0.390207	-0.52277	NA	-0.13742	0.243368	0.138911	NA	NA	NA	NA	-0.05441	8	-403.743	823.6698	3.323526	0.005841
0.374747	-0.52029	NA	-0.13879	0.240807	0.137452	NA	-0.14598	NA	NA	NA	8	-403.761	823.7065	3.360313	0.005735
0.390346	-0.52026	NA	-0.13901	0.240831	0.136372	NA	NA	NA	-0.00309	NA	8	-403.769	823.7215	3.375282	0.005692
0.387729	-0.55213	0.238954	NA	NA	0.252865	0.053748	NA	NA	NA	NA	7	-404.832	823.8074	3.461122	0.005453
0.388293	-0.54964	0.238562	NA	NA	0.26774	NA	NA	NA	0.038661	NA	7	-404.863	823.8695	3.523269	0.005286
0.379032	NA	0.216895	NA	NA	NA	NA	NA	NA	NA	NA	4	-407.914	823.8781	3.53191	0.005263
0.37073	-0.54948	0.232208	NA	NA	0.255972	NA	-0.14783	NA	NA	NA	7	-404.899	823.9418	3.595548	0.005098
0.386923	-0.54854	0.234188	NA	NA	0.255044	NA	NA	NA	NA	0.020814	7	-404.903	823.9496	3.603364	0.005078
0.371779	-0.49128	0.275265	-0.17759	0.298685	NA	0.077186	-0.16285	NA	NA	NA	9	-402.936	824.1027	3.756505	0.004704
0.389381	-0.49177	0.274703	-0.17687	0.297481	NA	0.077497	NA	0.014922	NA	NA	9	-402.941	824.1116	3.765387	0.004683
0.389012	-0.49213	0.273204	-0.17719	0.298117	NA	0.077006	NA	NA	-0.00981	NA	9	-402.943	824.1161	3.769833	0.004673
0.389455	-0.49084	0.276282	-0.17805	0.297741	NA	0.077679	NA	NA	NA	0.012619	9	-402.945	824.1194	3.773116	0.004665
0.382975	-0.44539	NA	-0.1045	NA	NA	NA	NA	NA	NA	NA	5	-407.041	824.158	3.811813	0.004576
0.385397	NA	NA	NA	0.195929	0.141807	NA	NA	0.044027	NA	NA	6	-406.056	824.2195	3.873231	0.004437
0.385693	NA	NA	NA	0.200224	0.134085	0.035502	NA	NA	NA	NA	6	-406.071	824.2486	3.902394	0.004373
0.384916	NA	0.254869	NA	0.213616	0.133532	0.056299	NA	NA	NA	NA	7	-405.053	824.2487	3.902456	0.004373
0.384696	NA	NA	NA	0.202408	0.137158	NA	NA	NA	NA	-0.05174	6	-406.08	824.2672	3.920916	0.004333
0.368133	NA	NA	NA	0.19961	0.136405	NA	-0.15728	NA	NA	NA	6	-406.095	824.2972	3.950985	0.004268
0.385086	NA	NA	NA	0.199293	0.13657	NA	NA	NA	0.000738	NA	6	-406.104	824.3151	3.968838	0.00423
0.384087	NA	0.247002	NA	0.208534	0.142542	NA	NA	0.042997	NA	NA	7	-405.09	824.3233	3.977087	0.004213

Table D.4: Model selection table for GLMM with probability of perceiving fair distribution of compensation.

(Intercept)	Compensation	Fishing dependence	Fisher association membership	Respondent identity	Awareness	Household income	df	logLik	AICc	delta	weight
-11.1138	21.05317	-0.56013	NA	NA	NA	NA	5	-192.404	394.885	0	0.206577
-10.9769	20.75322	-0.55416	0.752744	NA	NA	NA	6	-192.133	396.3727	1.487638	0.098185
-10.5378	19.79568	NA	NA	NA	NA	NA	4	-194.174	396.3986	1.513581	0.09692
-11.0933	21.0055	-0.55521	NA	-0.13674	NA	NA	6	-192.323	396.7541	1.869027	0.081139
-11.2649	21.41719	-0.56254	NA	NA	0.224493	NA	6	-192.349	396.8051	1.920114	0.079092
-10.8139	20.39828	-0.56678	NA	NA	NA	-0.03798	6	-192.394	396.8956	2.0106	0.075594
-11.3266	21.52562	NA	0.810324	NA	NA	NA	5	-193.862	397.8004	2.915356	0.048086
-10.962	20.72143	NA	NA	-0.15918	NA	NA	5	-194.062	398.2003	3.3153	0.039371
-11.0655	20.94467	-0.54863	0.775601	-0.15032	NA	NA	7	-192.035	398.2134	3.328349	0.039115
-11.1617	21.19592	-0.55722	0.78766	NA	0.270581	NA	7	-192.052	398.2474	3.362361	0.038455
-11.0187	20.87618	NA	NA	NA	0.185878	NA	5	-194.135	398.3465	3.461435	0.036597
-11.7061	22.35356	-0.56192	0.757929	NA	NA	-0.0438	7	-192.119	398.3817	3.496646	0.035958
-11.4158	21.71721	NA	NA	NA	NA	0.039732	5	-194.163	398.4019	3.516919	0.035595
-11.4605	21.84345	-0.55771	NA	-0.13625	0.222482	NA	7	-192.269	398.6814	3.796329	0.030954
-10.6192	19.96829	-0.55958	NA	-0.13269	NA	-0.02431	7	-192.319	398.7819	3.89687	0.029437
-11.2242	21.33076	-0.56987	NA	NA	0.228686	-0.04161	7	-192.337	398.8169	3.931918	0.028925

Appendix D references

DoF. (2014) *Hilsa fishing village statistics*. Dhaka, Department of Fisheries.

Appendix E: Supplementary material for Chapter 7

E.1 Conservation Trust Fund (CTF) Case Studies[40]

1. Mexican Nature Conservation Fund (FMCN)

Key informants: FMCN1; FMCN2

FMCN was established in 1994 as a non-profit civil association, following a three-year participatory consulting process funded by United States (US) Government agencies and a number of philanthropic organisations. It is one of the largest and most credible CTFs in the world, created with the aim of financing and strengthening efforts for the conservation and sustainable use of biodiversity in Mexico, through the operation of four conservation programmes: Protected Areas, Forests and Basins, Oceans and Coasts and Special Projects. Not only has it distributed over USD 65 million in support of nearly 1000 conservation projects, it has helped to promote sustainable businesses and public-private partnerships, built institutional capacity, and played a fundamental role in consolidating and improving Mexico's protected area (PA) system (Locker & Rosenzweig 2011). FMCN manages a total endowment of USD 120 million, capitalised by the US and Mexican Governments, the Global Environment Facility (GEF) and a group of US private philanthropic donors, which is complemented by a stream of earmarked sinking funds raised from diverse sources at a rate of USD 3-4 million per year.

FMCN is governed by a General Assembly composed of 32 members, representing both Mexican civil society and the international conservation community. The Assembly is responsible for approving annual reports and financial statements, and for appointing the Board of Directors, which provide the next level of authority. The Board is composed of 19 members from business and civil society organisations, selected from the Assembly members to meet the needs of the CTF and the diversity of its stakeholders, and including the Minister of Environment as an *ex*

[40] These case studies are published in Bladon et al. (2014). It should be noted that facts and figures were correct in 2013/2014.

officio member. In addition to strong leadership and good timing, the support of the Mexican Government from the beginning has been a key factor in success. At the same time, its autonomy has allowed it to last through three changes in administration. Due to the size and diversity of its board FMCN has been able to establish seven specialised technical committees to assist with the day-to-day operations of the CTF, including an Executive Committee. Asset managers were initially given control of financial strategy but, due to unsatisfactory results, they are now advised by an independent financial expert (Locker & Rosenzweig 2011).

Having established a successful track record in managing endowments, in 1996 FMCN was appointed manager of a separate endowment of USD 16.48 million from GEF, the Protected Areas Fund (FANP), established for the support of ten priority PAs. This is not a legal entity, but a fund which is segregated from FMCN's other programmes. It later received further capitalisation from GEF for the inclusion of 12 additional PAs in the programme, contingent upon 1:1 matching funds. FANP surpassed the required match, and today supports the management of 29 of Mexico's 176 PAs with an endowment of USD 76 million, many of which were previously paper parks. By covering basic operation costs of the PAs and building capacity, FANP has helped attract additional funds from both public and private sources. Furthermore, FANP has been able to customise disbursements to PA needs, making them available at a time of the year when Government resources are not – an example of diversified revenue sources strengthening each other's deficiencies.

2. Arannayk Foundation, Bangladesh

Key informant: AF1

The Arannayk Foundation, also known as the Bangladesh Tropical Forest Conservation Foundation, was established in 2003 by the Governments of Bangladesh and USA through a debt reduction program on interest payments, under the provisions of the US Tropical Forest Conservation Act (TFCA) of 1998. Its aim is to facilitate the conservation, restoration and sustainable use and management of tropical forests in Bangladesh, with key objectives including capacity-development of stakeholders and building partnerships between NGOs, the Government and private sector organisations engaged in activities related to forestry.

The CTF is registered as a non-profit company without shares under the Bangladesh Companies Act. Although these organisations are subject to more stringent government oversight than other NGOs registered under the Society Act, Arannayk is exempt from the usual government requirement for all foreign donations to pass through the Governmental NGO Bureau, a cumbersome and time-consuming process. Early on, little attention was given to fundraising and communication strategies, despite the fact that debt reduction funds were expected to be exhausted by 2018. But since the arrival of a new Executive Director in 2007, Arannayk has received USD 3.8 million from the World Bank, USD 1 million from GIZ (German Corporation for International Cooperation), financial and technical support from other development partners, and created an endowment (not yet tax exempt).

It is a small CTF, with only eight core members of staff and the Executive Director, appointed by a Board of Directors, which is comprised of representatives from USAID (United States Agency for International Development), the Government of Bangladesh (Additionality Secretary [Development] of Ministry of Environment and Forests), and five civil society representatives from relevant organisations such as educational institutes, NGOs, and the financial sector. Though it abides by Bangladesh Governmental policy, the Board has complete authority over grant-making decisions, using a panel of external experts to evaluate proposals. Arannayk started making grants in 2006 and by the end of 2015 had given over 93 financial grants and support to NGOs and other organisations active in forest and biodiversity conservation, and in 'exceptional circumstances' to the national Government. External evaluation has found Arannayk's policy and framework for administration and grant-making to allow effective and efficient operation meeting best practice standards, financial accountability and transparency (Mikitin et al. 2008). For example, Arannayk has begun delineating the priority areas and issues it wants to fund and calling for proposed implementation plans, which helps to focus efforts spatially and strategically. This call is followed by a peer review process and the proposals are chosen on the basis of a competitive scoring system.

3. Phoenix Islands Protected Area Conservation (PIPA) Trust, Kiribati

Key informant: PIPA1

The PIPA Trust was established under the Phoenix Islands Protected Area Conservation Trust Act of 2009 in the Republic of Kiribati, with the aim of providing sustainable financing for the protection of PIPA, one of the largest MPAs in the world. The Trust was founded by the Government of Kiribati, Conservation International (CI) and the New England Aquarium (NEA) as a result of a personal relationship between a representative of NEA and the President of Kiribati, whose strong interest in the conservation of Kiribati's natural resources opened conversations with Government officials which were critical to creating a trusting relationship.

In addition to some sinking funds (e.g. USD 1 million from GEF and support from regional governments) which are allocated to projects which require major capital expenditure, the PIPA Trust has established the PIPA Trust Endowment Fund (PTEF) in order to cover the costs of the Trust itself, the core costs of PIPA in accordance with the management plan and, most importantly, to finance the 'Conservation Contract' or PES arrangement between the PIPA Trust and the Government of Kiribati. Often termed a 'reverse fishing license', the arrangement compensates the Government for the fishing licenses which would have been sold to foreign fishing operations if the PIPA were not protected, in return for satisfactory performance under the contract. Though still in its infancy, PTEF has been capitalised with USD 2.5 million by CI and matching funds from the Government of Kiribati's Reserve Fund. Currently only 3.12 per cent of the MPA is no-take, but with further fundraising, PTEF aims to generate more compensation and thereby phase out commercial fishing (Davis 2013). By the end of 2014 it is expected that PTEF will be adequately capitalised to compensate for losses from 25 per cent no-take coverage, though the President of Kiribati is now calling for complete closure of PIPA, which would require about USD 25 million.

Key challenges in the creation of the PIPA Trust included the need to create a legal framework for marine conservation and CTFs, and the lack of scientific knowledge about the Phoenix Islands. However, strong public and political support was developed through publicity, which began with Kiribati's designation as a World Heritage site, relationship development with key Government officials, and the official status gained by establishing the PIPA Office within the environmental ministry, and hiring a senior bureaucrat. Without this unanimous and high level political support, the Executive Director noted that the donor community would not have come forward. Though the PIPA Trust Board of Directors is independent of the Kiribati Government, the Minister of the Environment joins representatives from CI and the NEA to form an equal management partnership. The project also benefited from Kiribati's previous positive experience with trust funds, and early relationships developed with New Zealand and Australia, which provided technical experience.

4. Banc d'Arguin Coastal and Marine Biodiversity Trust Fund (BACoMaB), Mauritania

Key informants: BACoMaB1; BACoMaB2

BACoMaB was established in 2009 as a foundation governed by UK law, with the aim of ensuring long term financing for Banc d'Arguin National Park (PNBA), a UNESCO World Heritage Site and the largest marine park in Africa. BACoMaB is partially financed through the EU-Mauritania Fisheries Partnership Agreement, an international PES scheme (Binet et al. 2013). In exchange for commercial exploitation of Mauritania's EEZ, the EU (European Union) pays Mauritania a fee, and since 2006 a proportion of this has been set aside to finance the operating costs of the National Park. In 2009, part of this annual payment was allocated to BACoMaB in the form of an endowment. The endowment is currently worth EUR 10 million, 2 million of which has come through the fisheries agreement, the rest from KfW Development Bank, MAVA Foundation and Tasiast Gold Mine through the Lundin Foundation. Commitments have also been made by AFD (French Development Agency) and FFEM (Fonds Français pour l'Environnement Mondial), and additional revenues will soon be explored, including revenues received from royalties received in connection with oil and gas exploitation concessions.

The idea of creating a CTF emerged in 2002, but the planning process was lengthy; it took time to get the Mauritanian Government on board, and enabling preconditions set out in a feasibility study had to be established, which involved a modernisation and restructuring of the PNBA authority. It was not until 2007 that a steering committee was finally put together by the main partners GIZ and the International Foundation of Banc d'Arguin (FIBA), for which a ministerial decree was required. Furthermore, although the Mauritanian Government agreed to earmark part of the Fisheries Agreement funds, initially this replaced their own contribution to the park, a situation which was rectified in 2008 (Binet et al. 2013).

A key factor in the successful establishment of BACoMaB was the support of FIBA, which helped to strengthen the management of PNBA and prepared an excellent communication strategy built on the notoriety of the park. FIBA also helped to lobby the Brussels Commission into supporting the Fisheries Partnership Agreement negotiations, which were in turn championed by the Director of the EU Common Fisheries Policy and which became instrumental in leveraging additional donor support. The mechanism has set a precedent and an attempt to replicate it is underway in Guinea-Bissau, where the CTF BioGuinea has formed an agreement with the EU which is being delayed by political constraints.

However, BACoMaB has experienced governance challenges. The appointment of an incompetent Director led to the suspension of some donor commitments this year. His poor leadership aggravated a lack of clarity amongst board members over roles and responsibilities, the need for which was underestimated by BACoMaB, leading to a division between Mauritanian and international members. Furthermore, annual revenue is eventually expected to guarantee effective management of PNBA, but given current market conditions funds would need to amount to EUR 100 million in order to generate substantial revenues, and by 2017 funds are not expected to exceed EUR 50 million.

5. Sangha Tri-National (TNS) Foundation

No key informants

The TNS Foundation was established in 2007 with a view to strengthening the long-term financing of the transboundary forest complex formed by three adjoining National Parks and their buffer zones in Cameroon, Central African Republic and Republic of Congo. Its creation began with the signing of a formal Cooperation Agreement to establish and manage the transboundary complex between the Governments of these countries in 2000. Although they differed in their initial aims and objectives, the formal links between the relevant ministries and the Foundation led to political endorsement which was crucial in its establishment (Klug et al. 2003; Usongo 2010).

As a multi-stakeholder and tri-national CTF, the process of creation was lengthy and expensive, with key support coming from WWF (World Wildlife Fund), GIZ, WCS (Wildlife Conservation Society), French Cooperation, and the Central Africa Regional Program for Environment (CARPE). A consultative meeting was held in which all actors involved in the management and financing of the TNS discussed the proposed plan and the composition of a steering committee (Klug et al. 2003). The selected steering committee then worked with experts in finance and a regional coordinator to bring together financial needs assessments for each TNS PA, which formed the basis of a business plan (Spergel & Taieb 2008; Usongo 2010). The Foundation is thus tailored to the specific needs of each National Park with four funding windows: one for transboundary management and one for each of the Parks.

Due to the financial insecurity and political instability of the TNS countries, the Foundation was registered as a UK charity, a decision which took more than 2 years for the TNS Governments to agree to (Usongo 2010).

At the time of writing, an endowment has been capitalised by KfW, AFD, and WWF Germany's Regenwald Stiftung, which channels funds through a targeted marketing campaign by Krombacher Brewery (TNS Foundation 2009). However, the Foundation is still a way off its minimum target of USD 22 million, and it has been suggested that TNS should therefore be exploring other mechanisms such as PES to secure the levels of funding required (Usongo 2010).

TNS Foundation is managed by an independent Board of Directors with representatives from the public and private sectors: the Governments of the TNS countries, WCS, WWF, Regenwald Stiftung, AFD and KfW (TNS Foundation 2009). These Directors have then nominated three private sector members from the TNS countries with expertise in conservation, law, business or finance. The Board is led by an Executive Director, based in Cameroon, and supported by two subcommittees specialising in fundraising and investment (Spergel & Taieb 2008). In 2009 TNS appointed an internationally recognised investment manager through an international tender process to manage funds according to Foundation's Investment Policy Statement.

6. Mesoamerican Reef Fund

Key informant: MAR1

The Mesoamerican Reef (MAR) Fund was created in 2004 in order to help finance the conservation and sustainable use of the marine and coastal ecosystems of the MAR region and the watersheds which drain into the Caribbean. By encompassing an entire ecoregion, it aims to consolidate and allocate donor contributions to common and strategic objectives in the area.

The importance of MAR is recognised by each of the four countries that share it, formalised through the signing of the Tulum Declaration in 1997 by the four heads of state. The MAR Fund relies on the technical, administrative and financial capacities of four pre-existing national CTFs in Mexico, Honduras, Belize and Guatemala. Though decisions are made by the Board of Directors, which includes representation from the four member CTFs, the Executive Director passes technical coordination to the member CTFs, allowing MAR Fund to benefit from their experience and save on costs. It was established with the endorsement of RedLAC and technical and financial support from WWF and TNC (The Nature Conservancy) as a US tax-exempt charitable foundation. Not only did this increase access to US donors who often require US charity status, it avoided navigation of the complex legislation and corruption in the participating countries.

MAR Fund launched its fundraising campaign in 2007, but the economic climate made it difficult even to raise project funds. Over 6 years it built a relationship with the German Government via KfW, and having proved that it could operate effectively at the regional level, obtained a five-year EUR 5 million grant for a project focusing on priority protected areas in 2010. Phase II of this project, again for five years and EUR 5 million, was formalised in 2013. In 2011, KfW also granted the Fund a USD 13.05 million endowment. Consequently, MAR Fund attracted additional investment (EUR 1 million) from the French Global Environment Facility (FFEM) in 2013 and a second contribution of EUR 7 million from KfW in 2014. It is optimistic about growth of the endowment, the revenue from which will now help finance operation costs and help to ensure a steady flow of finance to conservation initiatives in the region. Although it took longer than would have been desirable to raise the investment, this time has also allowed MAR Fund to incrementally strengthen its processes.

Since the growth of its endowment MAR Fund has expanded its grant-making strategy from a small grants programme and a community fisheries programme to operate larger projects focusing on a number of clearly defined priority PA sites. Requests for proposals are given with defined objectives, and rather than funds being disbursed equitably between the member CTFs, grants are made on the basis of competitive selection to ensure that funds are used as effectively as possible.

In 2013, MAR Fund committed to expanding their community fisheries programme, which runs in all four countries. It financially supports the active participation of self-organised groups of fishers in fisheries management and recovery, promoting an ecosystem-based approach which includes the establishment of critical fisheries recovery sites (no-take zones, fish refuges and community marine reserves). For example, in Honduras, three communities organised themselves to create an association and signed an agreement with the co-administrator of an MPA for co-management of a fishing territory with no-take zones. They designed and monitor these together, and they identified training and capacity needs for which they receive external financial assistance from MAR Fund. In Guatemala, fisher associations organised themselves to establish no-take zones, and although they work with the Government agency that administrates the PA in a 'discussion table' for the no-take zones, they undertake control and surveillance measures on their own, in response to the lack of financial resources within Government for these activities. In a newly established no-take zone in Honduras, close to the Guatemalan border, seven communities have been supported to establish the site, do a baseline analysis of the local fisheries and to monitor it. In Guatemala and Honduras associations do not have formal fishing rights over their managed areas, but in Belize MAR Fund has supported the implementation of fishing rights in one PA, and in Mexico fishing cooperatives can legally obtain fishing concessions.

7. Caribbean Biodiversity Fund

Key informants CBF1; CBF2

The Caribbean Biodiversity Fund (CBF) is a regional endowment fund, launched in 2012 with the support of TNC, GEF (through The World Bank and United Nations Development Project [UNDP]), and the German Government through KfW, who pledged USD 42 million. The CBF, together with eight national level CTFs (in Antigua and Barbuda, the Bahamas, Dominican Republic, Grenada, Jamaica, St Lucia, St. Kitts and Nevis and St Vincent and the Grenadines), form the Caribbean Sustainable Finance Architecture for Conservation. This Architecture is being designed and established as a vehicle to achieve commitments made by the participating countries under the Convention on Biological Diversity and the Caribbean Challenge Initiative, to support national environmental priorities and to close the financing gap for conservation.

By 2016, the CBF aims to start disbursing funds to some of the eight proposed national level CTFs, and if successful it will be the first regional endowment in the world to support multiple national level Trust Funds for marine and terrestrial PA management (TNC 2012). This design decision was made on the basis of economies of scale; investing the donor capital as one endowment fund made more financial sense than investing it in eight separate funds. The hope is that the CBF will over time attract additional donor capital, allowing different thematic windows to be opened for growing the endowment.

Legal agreements will be set out between each national CTF and the CBF before disbursements can be made. Negotiations are already underway with the CTF in the Dominican Republic and, at least, three additional negotiations are expected to be underway in 2016. The CBF is also exploring and assisting CTFs in identifying, designing and implementing sustainable mechanisms for the generation of matching funds by each participating country. These matching mechanisms can be established through government instruments, private sector, or public-private partnerships. Although the initial focus was on generating revenue through government instruments, due to the current economic climate the CBF is now considering more creative and less government-centric financing tools. It is also important to note that CBF disbursements must supplement, not replace, current government financing for the environment.

The CBF currently has two members on its Board of Directors, representing KfW and TNC. As each national CTF signs a legal agreement with the CBF, it will be able to nominate a Director from its Board to be part of the CBF Board. The only restriction is that at all time the CBF Board must have a non-government majority, which the CBF hoped would be an incentive to sign early on. In this case, creating the first agreement quickly will be a key factor in success of the CBF; there is a competitive instinct among the islands.

Five of the CTFs expected to become CBF partners are already legally established (Antigua and Barbuda, the Bahamas, Dominican Republic, Jamaica, and Saint Vincent and the Grenadines). One (Dominican Republic) is already negotiating the terms of the agreement with the CBF and the other 4 are eagerly working on preparing the necessary documentation to officially become

CBF eligible. Three other CTFs have advanced significantly their legal establishment. CTFs are being established through the Company's Act or legislation. Most countries have opted to use the Company's Act. According to the Chief Executive Officer, one of the greatest challenges in the process for establishing this regional financial architecture has been the political process and working through the establishment of nine CTFs all at the same time. CTFs are not a traditional institution in the Caribbean, and finding the balance between independence from government and supporting national agendas has required intensive policy and technical work. In addition public-private partnerships in the Caribbean on environment issues are still an emerging way to address environmental priorities, and different sectors have reservations about how these partnerships can be effectively managed and deliver strong results. The previous Executive Director noted that, in hindsight, more effort could have been put into sensitising influential people at an earlier stage. In addition, the inevitable bureaucracy inherent in an initiative of this size, involving so many stakeholders, also results in process delays.

However, commitment from governments and other stakeholders continues to be high and significant progress has been achieved. The CBF and national stakeholders have put a lot of effort and resources into the technical development of the CTFs and sustainable finance mechanisms. In the near future the CBF will be establishing a multi-year Strategic Plan to consolidate the Caribbean Sustainable Finance Architecture for Conservation.

8. Bhutan Trust Fund for Environmental Conservation (BTFEC)

No key informants

BTFEC was established in 1991 by the Royal Government of Bhutan with financial support from WWF and technical assistance from UNDP, in order to provide sustainable financing for environmental programmes in the country and allow the national treasury to focus more on direct poverty reduction (Namgyal 2003). It received USD 10 million from the GEF in 1992, and went on to raise additional capital from the Governments of Bhutan, Denmark, Finland, Netherlands, Norway and Switzerland (GEF 1998). The original capitalisation of just over USD

21 million has since risen to USD 44 million. Although it was let down by an initial inappropriately low risk investment strategy, the success of BTFEC's fundraising has been attributed to a combination of pre-existing political will, links with aid agencies and the disbursement of GEF contributions in tranches which were conditional on, among other factors, securing other donor contributions (GEF 1998).

BTFEC is governed by the 1996 Royal Charter of the Trust Fund for Environmental Conservation and managed by a board of seven Bhutanese members, including the Director who acts as an *ex-officio* Member Secretary. Five of these members are government-appointed, and include the Minister of Agriculture and Forests as the chairman, the Secretary of the National Environment Commission, the Director of National Budget, Head of Policy and Planning in the Ministry of Works and Human Settlement. The non-governmental representative is the Executive Director of a national NGO, though the board once also included two donor representatives. The Director chairs a seven-member technical advisory committee, and the board is also advised by an asset management committee. Board decisions are implemented by a small secretariat of staff. BTFEC has tax-exempt status with the US Government and assets have been managed by a professional asset manager based in the US since 1996 (Namgyal 2003).

Much of BTFEC's support has been directed towards building institutional and human capacity; this was Bhutan's largest constraint to conservation, and donors like to see their investments yielding immediate results (Namgyal 2003). It has strengthened the absorptive and implementation capacity of government agencies and played a role in the creation and expansion of Bhutan's first NGOs (GEF 1998; Klarer & Galindo 2012). This success may probably contributed to the subsequent creation of more CTFs around the world. However, there are areas of BTFEC's operation which require work, including the monitoring and evaluation of biodiversity impacts, diversification of revenue sources, and grant-making procedures; it has been criticised for a lack of clear focus at an early stage (GEF 1998).

9. Yasuni-ITT Trust Fund

No key informants

The Yasuni-ITT Trust Fund was established in 2010, managed by the Multi-Partner Trust Fund Office of UNDP and to be capitalised under the Yasuni-ITT Initiative, which was launched by the President of Ecuador. Under this initiative, the national Government proposed to forego exploitation of the Ishpingo–Tambococha–Tiputini (ITT) oil fields, which lie under the core of the Amazonian Yasuni National Park, and are thought to be one of the most biodiverse parks in the world (Bass et al. 2010). In exchange, it asked for USD 3.6 billion in public and private donations from the international community over a 13 year period, the interest from which would accrue in the Trust Fund for use in funding national sustainable development projects (Larrea 2009). In return, the Government would issue guarantee certificates reflecting the carbon value of contributions, tradable on the EU carbon credit market (Finer et al. 2010). The initiative therefore promised not only to result in the protection of biodiversity and indigenous territory, but by preventing deforestation and locking away over 850 million barrels of crude oil, it would also prevent emissions and thus address climate change.

In the years that followed, this compensated moratorium was described as 'innovative' and 'precedent-setting'; it garnered support from the Ecuadorian public, from Germany and the EU; received endorsements from high profile figures including UN General Secretary Ban Ki Moon; and the UNDP even suggested that it might serve as a model for conservation around the world (Larrea 2009; Finer et al. 2010; Marx 2010). But despite pledges from governments, NGOs and individuals, only USD 13 million was deposited and by August 2013 the liquidation of the Trust Fund was announced (Petherick 2013).

Perhaps the failure of donors to pay up was largely a product of the financial crisis (Petherick 2013), but there were those who suspected failure from the beginning. Economists criticised the estimation of lost revenues, citing flawed valuations and unclear accounting (Finer et al. 2010; Haddad 2012). Others blame the fundraising process; its compensation framework essentially shifts the burden of responsibility from the state to the international community, which undermines the philosophy behind UNFCCC and REDD+ (Haddad 2012). There was also a great deal of concern over the integrity of the Ecuadorian Government and resentment for the way it handled the negotiation process, which has been termed 'environmental extortion' (Petherick 2013). Multiple changes in institutional design bred scepticism (Arsel & Angel 2012), and not only did the President continue to pursue the licensing procedure for oil extraction in case of initiative failure, there was nothing to stop Ecuador reneging on its promise to forego drilling after payment and no guarantee that funds would be returned if this happened (Petherick 2013). A further factor was the political instability of Ecuador and subsequent lack of trust from the outset (Finer et al. 2010). But the greatest flaw appears to be governance; it was clear from its inception that the Yasuni-ITT initiative was to be driven by the Ecuadorian Government itself, despite promises of increased engagement with civil society (Arsel & Angel 2012). The proposal was in fact rejected at the Copenhagen Climate Summit in 2009 because the Government of Ecuador wanted more authority over the fund (Marx 2010). Discussions were conducted at an institutional level, civil society and particularly indigenous populations were largely excluded from the decision-making process, and the Board of Directors was government-dominated (Arsel & Angel 2012). Ironically, the initiative was born outside of the state, but those who had initiated and been involved in early discussions were soon offered state positions and continued their involvement not as representatives of civil society but of the Government (Arsel & Angel 2012).

10. Fund for the Protection of Water (FONAG), Ecuador

No key informants

FONAG is a water conservation fund and public-private partnership established in Quito, Ecuador, in 2000 (Benítez et al. 2009). Quito derives most of its water flows from PAs in the Andean region, so FONAG was created with the aim of pooling demand for watershed services among its various beneficiaries through PES, in order to improve Quito's water quality and quantity in the long-term. It is an 80 year endowment fund, regulated under Ecuador's stock market law, and launched by TNC, local NGO *Fundación Antisana*, the Mayor of Quito through the Quito water utility, and USAID with a seed capital of USD 21,000 (RedLAC 2010). In the last decade it has received contributions from Quito's electrical utility, a private brewery and a water bottling company, returns from which have allowed it to leverage matching funds from international and local NGOs and governments, and has grown to an endowment of over USD 6 million (RedLAC 2010). 80 per cent of annual returns are invested in conservation projects within the watersheds that supply the city, which by 2008 had amounted to USD 9.3 million (RedLAC 2010). All contributions were initially voluntary, but in 2006 an ordinance was passed which established a mandatory 2 per cent contribution on fees collected by the water utility, increasing long-term financial stability of the fund without increasing water user fees (Goldman-Benner et al. 2012).

FONAG was the first water fund to be established and has since served as a model for a number of others in Ecuador and throughout the Andes region, which lends itself to the creation of water funds (Goldman-Benner et al. 2012). Although FONAG was criticised in the early days for making very small grants to conservation projects despite large amounts of money coming into the fund (Benítez et al. 2009), it has nevertheless made enormous contributions to watershed conservation. There is evidence that FONAG has reforested 2,033 ha of land with 2 million trees, engaged children in environmental education programmes and families in community development projects, trained and employed community-based park guards, and maintained ecosystems in a pristine state, and ensured sustainable and transparent use of resources (Goldman-Benner et al. 2012). A major factor in its success was the willingness of beneficiaries to invest from the outset, despite legal challenges, political instability, and a lack of clear science linking watershed services with downstream benefits (Echavarria 2002; Postel & Thompson 2005). This was in part due to excellent communication of the benefits to all stakeholders from the start. For example, it produced a short publication which was also used to raise awareness amongst the city's residents and to gain support from the Mayor's office – support which was critical due to its authority over the municipal water utility (RedLAC 2010). It also clearly established the activities for which the funds would be used, identifying five watersheds as priority areas.

In 2004 FONAG hired a high profile Technical Secretariat with expertise in watershed management, which strengthened institutional capacity. The Secretariat led a strategic planning process to develop FONAG's projects, with approval of the Board of Directors, which includes representatives from local communities; local NGOs, government; the national PA authority; and business (Benítez et al. 2009). Funds are invested by an independent financial manager (Postel & Thompson 2005).

11. The Protected Areas Conservation Trust (PACT), Belize

Key informant: PACT1

PACT was established as a public-private CTF in 1996, under special legislation, with the aim of financing the management of Belize's extensive PA network. Initial capital was provided by USAID (USD 72,000), but PACT is primarily financed through a revolving fund generated through tourism taxes or conservation fees, including a 20 per cent commission on cruise ship passenger fees and a USD 3.75 visitor departure fee. 5 per cent of total revenues are now deposited in an endowment fund, which would only be used in extreme circumstances and is currently approximately BZUSD 6 million. PACT also manages a sinking fund created through a small debt-for-nature swap with the US Government under the TFCA.

Although the bill allowing for the creation of the CTF had already been drafted by 1992, the stakeholder consultation process was very lengthy and it was another five years before PACT became operational; a delay which was largely a result of lengthy negotiations with the tourism sector. There was concern that the size of the fees on top of other taxes already in place would discourage tourists, and concern over a proposed government majority on the Board of Directors. A general election and lack of interest from the new ruling party also delayed the bill.

Negotiations were eventually successful and PACT is now governed by a non-governmental majority Board of Directors with representatives from all stakeholder groups, including an independent finance expert. PACT ensures that it complements governmental funding, rather than supplementing it, by ensuring that grants are awarded based on conservation criteria evaluated by a technical advisory panel which makes recommendations to the Board. Furthermore, PACT legislation prohibits the funding of salaries or any recurring expenses of government agencies or NGOs, or any profit-making organisation; although NGOs have been lobbying for PACT to support a portion of their core costs as donor contributions diminish, and a clause allowing for the support of up to 20 per cent of NGO core costs is currently under negotiation.

PACT started out as a small grants programme and has since expanded to give grants of ten types which fall under two broad categories – project grants and capacity-building grants – and gained an international reputation for its strong policies and structures. However, PACT is in a state of change after what the Senior Grants Officer refers to as a period of 'stagnation'. Following an institutional assessment completed in 2011, PACT is now pursuing growth more aggressively and looking to expand its role in conservation. For example, it is in the process of creating a private PACT Foundation which will be able to raise additional donations inaccessible to the main CTF due to government involvement. If this had been achieved earlier, perhaps PACT would have made more progress in filling the USD 6 million-gap in funding which is needed to sustainably manage the country's PA network. The failure to recognise the need to diversify PACT's funding sources earlier may be in large part due to the lack of formal assessment of the PA network until recently; it is only now that PACT has begun working with the government on a national PA system and policy that it has been able to identify needs at the system level and begun exploring additional modes of fundraising. Furthermore, PACT is working with RedLAC to pilot a monitoring and evaluation methodology which incorporates biodiversity impacts more systematically – impacts which have really only been assessed on an ad hoc basis in the past.

12. Fondo Accion, Columbia

Key informant: FA1

Fondo Accion, also known as The Fund for Environmental Action and Childhood, has origins in a bilateral agreement between the US and Columbian Governments in 1993, which created the Enterprise for the Americas Initiative Account (EIA) and channelled USD 41.6 million of capital into a fund through a debt reduction agreement. Fondo Accion itself was established in 2000 as a private NGO, and took over the administration of the EIA account from a failing NGO platform, with the aim of co-financing projects intended to protect and sustainably manage Columbia's natural resources whilst promoting child development. In 2006 the account was divided into a sinking and endowment fund in order to increase financial sustainability; the sinking fund has since been depleted, and the endowment fund, now USD 33 million, has been used to successfully attract new donors, establish new sub-accounts for specific purposes, and has allowed Fondo Accion to grow substantially in size and permanence.

Furthermore, in 2004 its bylaws were modified, allowing Fondo Accion to manage other additional accounts, and a debt-for-nature swap was signed under the TFCA by the same governments with TNC, WWF and CI, generating another USD 10 million over 10 years for use by NGOs and Community-Based Organisations with previous experience in tropical forest conservation in the targeted areas. Some of this is managed as an endowment and used to fund the management of PAs, buffer zones and corridors, while the remainder is a sinking fund used to provide direct funding for sustainable development and conservation activities.

The Directive Council or Board of Directors makes decisions and recommendations through an Executive Secretary who oversees the Executive Secretariat, a team of 21. The TFCA account is managed by a separate Oversight Committee. Financial management is overseen by a financial commission of three members, and performance is monitored by the Directive Council and Executive Secretariat. The Board has always comprised of diverse representatives; one from USAID, two from the Columbian Government, and five from civil society. The incorporation of representatives from the corporate sector has been very valuable in terms of strategic planning, and contributed to growth of the CTF.

A change of directorship in 2004 led to a number of improvements in the operations of the CTF. There had previously been very little documentation of procedures, but he introduced a system of organisation, created trust within and between the board, introduced a legal area and an organised accounting system, and improved communication generally. An important factor in success of the CTF is the existence of a strategic plan for which implementation is carefully monitored. Fondo Accion is one of only two Latin American CTFs to have its own quality management standard, similar to that which most other private enterprises use, which standardises processes and has been particularly useful for the arrival of new members.

E.2 Key informant interviews

Key topics in general study key informant interviews:

1. Rationale for creation
2. Planning
3. Legal framework
4. Governance structure
5. Fundraising
6. Administration and finance
7. Grant-making
8. Monitoring and evaluation
9. Impact
10. Challenges and setbacks
11. Conditions for success
12. Payments for Ecosystem Services

Key topics in Bangladesh case study key informant interviews[41]:

1. Constraints and limitations of current hilsa fisheries management
2. Strengths of current hilsa fisheries management
3. Opinion of the compensation scheme and impact on the hilsa fisheries and livelihoods
4. Impact of current management on livelihoods
5. Opportunities for improved hilsa fisheries management
6. Trust Funds and PPPs in Bangladesh
7. CTFs in Bangladesh
8. The prospect of setting up a hilsa CTF

[41] Questions 1-5 were largely the focus of chapter 5 and were used minimally in this chapter.

Appendix E references

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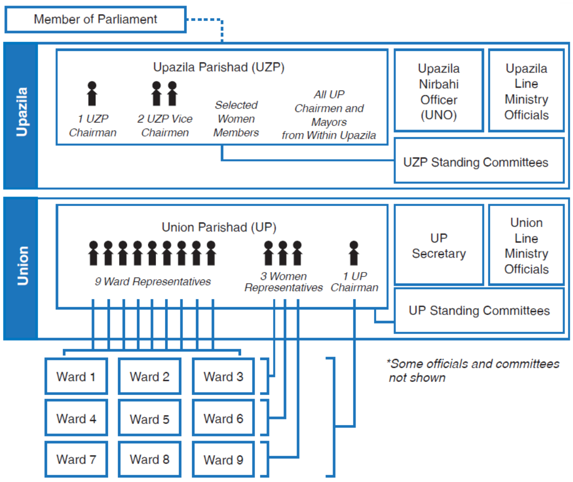

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