Understanding fishers' spatial behaviour to estimate social costs in local conservation planning

ANDREA PAULINE COOMBS WALLACE

B.Sc., Trent University, 1994
M.Sc., University of British Columbia, 2004

A THESIS SUBMITTED FOR THE DEGREE OF

DOCTOR OF PHILOSOPHY

IMPERIAL COLLEGE LONDON

Centre for Environmental Policy / Division of Ecology and Evolution

October 2012

Declaration

This thesis is the result of my own work. The work of all others is appropriately acknowledged and referenced in the text.

Andrea Pauline Coombs Wallace, 2012

Abstract

Artisanal fisheries are a key source of food and income for millions of people worldwide. However, unmanaged or excessive fishing activity can lead to declining returns for fishing effort and livelihood insecurity, and adversely impact wetland ecosystems. Management interventions such as protected areas and temporal closures may improve fishery sustainability and reduce environmental degradation, but often carry costs for fishers. Understanding predictors of fishing behaviour would allow conservation planning to minimise the adverse impacts of interventions, increasing the likelihood of fisher support for change. However, factors influencing fishers' behaviour are rarely identified or taken into account when implementing conservation actions.

Madagascar's Lake Alaotra wetland supports the nation's largest and most productive artisanal freshwater fishery, and provides critical habitat for endemic wildlife. Local fishers depend on the fishery for livelihood throughout the year. Catch-monitoring interviews, semi-structured interviews, focus groups, and follows were conducted over 16 months with 784 fishers at Lake Alaotra to understand the socioeconomic dynamics of the fishery. Although information from fishers was sometimes imprecise, participatory monitoring methods engaged fishers and improved understanding of system dynamics. Linear mixed models confirmed that proposed restricted areas and temporal closures would generate direct short-term costs through reduced catch sizes, which vary between gear types. Socioeconomic data, spatial distribution of fishing effort, and fishers' evaluations of management scenarios were used to explore alternative strategies. The conservation planning tool Marxan was used to identify reserve networks capable of achieving conservation goals while minimising adverse impacts for fishers.

The research demonstrates that: interventions can have unequal impacts on local people; information about costs and benefits of interventions can produce more realistic and implementable conservation plans; and actively engaging fishers and understanding their spatial behaviour at relevant scales is critical for managing fisheries sustainably and promoting effective long-term conservation of freshwater ecosystems.

Acknowledgements

Firstly, many thanks to my supervisors, Professor E.J. Milner-Gulland, Dr Emily Nicholson, Dr Julia Jones, and Dr Richard Young for your unwavering support, guidance, and patience. Your thorough and constructive comments on my work have greatly improved my critical thinking and this thesis. Each of you has been an inspiration!

I also thank the Durrell Wildlife Conservation Trust and the Economic and Social Research Council of the UK for providing financial support for this research.

The Imperial College Conservation Science (ICCS) lab is an amazing research group and my PhD experience would not have been as enjoyable without them. Many thanks to Emily, Sarah, and Ana for being great office mates, Tim and Joe for the frequent "coffee/hot water?" calls and for reminding us when lunchtime was approaching, and Nils for always being entertaining and super helpful with stats! I will miss the 'core group'. Thanks also to all ICCS predecessors (Aidan, Matt, Caroline, Kerry) and successors (Hanna, Henry, James) for your friendship and great discussions. Thanks also go to the Silwood accommodation office for giving us a home after fieldwork.

Many thanks to the Durrell team in Madagascar, particularly Bary-Jean Rasolonjatovo and Jonah Ratsimbazafy who provided invaluable orientation and support. Lala Jean Rakotoniaina was a key source of local information and greatly assisted me with translations from Malagasy to French as well as explaining the traditional Anororo calendar. Hasina Randriamanampisoa sorted out all our administration, permit, and visa requirements. I thank the rest of the staff at the Durrell office in Antananarivo (Richard, Aina, Ony, Juliette, Lynda, Emma, Lance, and Herizo) for welcoming Graham and I to Madagascar and making us feel instantly like part of the team.

I also thank the staff of the government institutions in Ambatondrazaka who kindly gave their time to meet with us when we first arrived, as well as on several occasions thereafter, including staff from the Service Régional de la Pêche et des Ressources Halieutiques (Fisheries Service), the Ministère des Eaux et Forêts (Ministry of Water and Forests), and the Fédération des Pêcheurs (Federation of Fishers), as well as the

Department of Environment & Forest (DREF) for approving our research permits. Many thanks to M. Jarrel Rabemazava for providing access to the library at the Fisheries Service as well as offering his assistance when needed. I am particularly grateful to the Mayor, Simon Mosahalison, and Deputy Mayor, Jean Desiré, of Anororo for welcoming Graham and I into Anororo village, granting permission to conduct this research, and for their interest in the study.

Joachin Randriarilala, Solofoniaina Esperant Rakotonisainana, and Luhanaud Andriamiarivola were extremely helpful, knowledgeable, and hard-working research assistants who ensured the project ran smoothly and was a success in the field. Many thanks to M. Rabemanisa, our tireless paddler and key informant, who always managed to keep the canoe steady despite us, and to Narcisse and Ravo Andriamizana who also helped with paddling and data collection. Immeasurable thanks go to our hosts in Anororo, Mme Aimee and M. Robin, who took us into their home and always looked after us. I will always remember our dinner conversations rifling through our English-Malagasy dictionaries to understand what we were trying to say to each other, or teaching Fensoa numbers and letters in English. I am very grateful to Rado Zilia Randriamihamina for arranging our host family, finding research assistants, and helping with data collection when time permitted. Thanks also go to Mme Eleonore for all our pharmaceutical needs.

I am extremely indebted to the fishers and community of Anororo village for their participation, support, and enthusiasm for the project. *Misaotra betsaka* for your friendship and for sharing your knowledge and time.

And above all ...

Thank you to my father, J. Allen Coombs, for his inspiration and confidence in me. This one’s for you Dad!

And to Graham, my pillar of strength and support throughout. Thank you for being you.

Table of Contents

List of Figures

List of Tables

CHAPTER 1 INTRODUCTION

1.1 PROBLEM STATEMENT

Conservationists increasingly recognise that resource management strategies must account for the needs of local people in order to be successful, particularly over the long term (Newmark et al., 1994; Ticheler et al., 1998; Welcomme, 2001; Danielsen et al., 2005). This is largely because local people often bear direct and opportunity costs from conservation actions. An extensive body of research highlights the importance of resource users in system dynamics (e.g., Smith et al., 1999; Sainsbury et al., 2000; Ostrom, 2009; Pressey and Bottrill, 2009). Although those involved in conservation planning increasingly consider social dimensions, the potential costs of management actions on resource users are often not fully understood (see Hulme and Murphree, 2001; Campbell and Vainio-Mattila, 2003; Weladji et al., 2003). Furthermore, a large proportion of management strategies fail to be implemented, or are poorly implemented, due to gaps between research and action (Knight et al., 2008). In developing countries this gap often arises due to a mismatch of scales, with planning typically occurring at regional scale and implementation occurring locally (Weeks et al., 2010b). It is also imperative to extend the role of resource users beyond their influence on system functioning to include their impact on governance and potential participation in management decisions (Anderies et al., 2004; Brondizio et al., 2009).

Inland fisheries are an important source of food and income for subsistence and artisanal communities in many areas of the world (Welcomme, 2001; Canonico et al., 2005), and provide illustrative examples of the challenges involved in managing natural resources. Although living aquatic resources are renewable, mismanagement and overexploitation can lead to loss of fish biodiversity, declining catches, poor returns for fishing effort, and livelihood insecurity (Amarasinghe and de Silva, 1999; Welcomme, 2001; Mugisha and Ddumba, 2007). Fishing activity can also adversely impact adjacent wetland habitat, undermine ecosystem integrity and functioning (Leal, 1998; Christensen and Pauly, 2004; Walters, 2004), and thereby compromise conservation efforts. Management interventions such as aquatic protected areas and/or closed seasons can improve the sustainability of fisheries and reduce environmental degradation over the long-term (Roberts et al., 2001; Cucherousset et al., 2007); however, they can also involve short-term costs to fishers by restricting areas that are open to fishing and potentially reducing catches (Ban and Klein, 2009). These costs may make compliance impossible for fishers whose livelihoods depend on day-to-day catches, and consequently the longer-term goals of management may be compromised (Peterson and Stead, 2011). It is therefore necessary to mesh ecological goals with the interests, concerns, and livelihoods of local stakeholders to develop effective long-term conservation strategies that are socioeconomically viable. This requires an interdisciplinary approach (Reyers et al., 2010) that acknowledges the key role of fishers in determining the sustainability of aquatic protected areas while incorporating an assessment of the livelihood impacts of conservation for those fishers.

An improved understanding of artisanal fisheries, fisher perceptions and behaviour, and the impacts of conservation actions for those that rely on fisheries resources is fundamental for effective participatory resource management and planning (see Heemskerk et al., 2003). However, spatially-explicit information about resource-user behaviour and the costs of conservation actions for resource users is frequently lacking or poorly understood and rarely quantified when designing and implementing management interventions (Redman, 1999; Margules and Sarkar, 2007; Levin et al., 2009). Similarly, there is often a lack of knowledge about how governance systems may impact resource users, and how resource users could contribute to effective governance procedures (Anderies et al., 2004; Winkler, 2011). Incorporating the dynamics of costs and resource-user behaviour into conservation planning is complex, and to date has not been achieved in a meaningful or effective manner (Stewart and Possingham, 2005; Balmford and Cowling, 2006). In particular, the behaviour of subsistence or artisanal resource users, and the costs of conservation for these users, are rarely specified or quantified in sufficient detail for accurate assessment of the impacts of management interventions (Ban and Klein, 2009).

Lake Alaotra, Madagascar is a particularly suitable site for investigating the issues and interrelationships outlined above. The Lake Alaotra fishery is the largest and most productive freshwater fishery in Madagascar (Andrianandrasana et al., 2005), and artisanal fishing occurs throughout the lake and adjacent marsh (Pidgeon, 1996). The fishery is regulated by a multi-level governance system that has failed to adequately incorporate socioeconomic information or consider resource-users' views, incentives, and costs. Enforcement of regulations has been sporadic and inconsistent. The fishery is within an area of significant conservation value that has been declared a Ramsar site to protect biodiversity, including the critically endangered and endemic Alaotran gentle lemur (*Hapalemur alaotrensis*) (IUCN, 2010). Conservation of the marsh and lake directly interlinks with management of the fishery as well as fisher behaviour. However, despite an extensive history of fish introduction and re-stocking, systematic efforts to manage the fishery for the ongoing livelihood benefit of local people and conservation of wetland habitat are relatively recent (Durbin et al., 2003). The fishery continues to be under great pressure and requires effective management to promote conservation (Durbin et al., 2003). It is an objective of the Malagasy Government and local communities to ensure the fishery is sustainable, protect the wetland habitat and biodiversity, and thereby improve the livelihood of local fishers (Andrianandrasana et al., 2005).

Using Lake Alaotra as a case study, this research is an investigation of the fishing behaviour of artisanal fishers, and costs to fishers of conservation and management actions. An interdisciplinary approach was used to examine and quantify the impacts of fishery management and conservation strategies from the perspective of fishers as resource users within a social-ecological system. Although various spatial, temporal, and gear-based fishing restrictions have been implemented at Lake Alaotra over the past decade, these measures have not been effective. This is probably partly because compliance involves costs for fishers, who have not been able to participate fully in the planning process to ensure their interests are taken into account. Adopting a spatially explicit approach within a conservation planning framework, the research informs approaches to allow information from and about local people to be integral to planning. This will enhance conceptual understanding of the spatio-temporal dynamics of systems subject to conservation planning and natural resource management, and improve methods for taking these dynamics into account in conservation interventions; results from this research will therefore be widely applicable beyond the case study.

The study extends the work of the Durrell Wildlife Conservation Trust (DWCT) in villages around Lake Alaotra. Practical applications include assessment of the status of the Lake Alaotra fishery for ongoing monitoring, an understanding of fishing effort and the spatial distribution of fishing, quantifying costs to fishers of alternative conservation interventions, identifying more-efficient reserve networks, and engaging local people in conservation and resource management issues. Given that wetland habitats have traditionally been overlooked when investigating fisheries issues (Ratner et al., 2004), the research also provides an important opportunity to better inform policy and strategy decisions to manage an internationally recognised and locally important wetland impacted by a relatively large artisanal fishery.

1.2 AIM AND RESEARCH OBJECTIVES

The aim of this research is to gain an understanding of the spatial behaviour of fishers as well as the spatio-temporal dynamics of the costs to fishers of current and potential conservation and management. The research aim will be addressed using a case study of an inland artisanal fishery in Madagascar (Lake Alaotra) through the following objectives:

1. Characterise the Alaotra fishery to gain an understanding of system dynamics;
2. Analyse the drivers of spatial and seasonal fishing behaviour, as a function of revenues, costs, and constraints;
3. Assess the costs for fishers of alternative conservation interventions, based on a range of assumptions about fisher responses, using spatially-explicit data and management scenarios;
4. Integrate spatially-explicit cost information into conservation planning for the Lake Alaotra system, and use this to make explicit the potential costs of restricted areas.
5. Make recommendations for conservation and management of the Lake Alaotra system, in light of fishers' perceptions and behaviour, in order to enable improved fisheries management.

1.3 THESIS OUTLINE

Subsequent to this introductory chapter, the thesis has the following structure:

Chapter 2 – Research background & conceptual framework

This chapter provides a review of relevant literature and the conceptual framework for the research.

Chapter 3 – Study site & research methods

This chapter provides a detailed description of the study site including information about the fishery, local governance, and conservation efforts at Lake Alaotra, as well as an overview of the research methods and data-collection techniques used during the study.

Chapter 4 – Characterising the Lake Alaotra fishery: Implications for management

This chapter develops a comprehensive understanding of the Anororo fishery, including fisher demographics, fishing locations used, fishing methods used, catch characteristics, and the socioeconomic context of fishing activity, as well as qualitative and quantitative indicators of the state of the fishery.

Chapter 5 – Predicting the potential impacts of conservation interventions on fishers

In this chapter I use linear mixed effects models to identify the drivers of catch size and explore the potential effects of spatial and temporal interventions for different groups of fishers.

Chapter 6 – Understanding the drivers of fisher effort and spatial behaviour at Lake Alaotra, Madagascar

This chapter uses information from semi-structured interviews, focus groups, and catch interviews with fishers to understand why fishers fish where they do, and the factors influencing the amount of time they invest in fishing.

Chapter 7 – Artisanal fishers' perceptions of management interventions: Using scenarios to understand how fishers would respond to change

This chapter draws on information from semi-structured interviews and focus group sessions to understand fishers' perceptions of the potential impacts of regulations on their fishing behaviour. Scenario analyses explore how fishers may change behaviour in response to management interventions and changing conditions.

Chapter 8 – Including fishers in conservation planning: The costs of alternative reserve networks

In this chapter I develop potential reserve designs for the Lake Alaotra wetland that meet biodiversity targets while also minimising costs to local fishers. I compare new reserve configurations with the current management plan and identify priority areas for conservation.

Chapter 9 – Discussion

This chapter provides a synthesis of the findings of the research, key implications for conservation and management of social-ecological systems, policy recommendations, and directions for future research.

CHAPTER 2 RESEARCH BACKGROUND & CONCEPTUAL FRAMEWORK

2.1 LITERATURE REVIEW

2.1.1 Social-ecological systems

All natural resources used by humans occur within complex, multiple-level social-ecological systems (SESs; Ostrom, 2009). The 'ecological' (or biological) and 'social' (or human) components of these systems interlink and influence each other through interactions across a range of spatial and temporal scales (Berkes, 2003; Anderies et al., 2004; Berkes, 2004; Ostrom, 2007b). The core components or subsystems of a SES, together with the primary links between them, are illustrated in a framework developed by Ostrom (2007a, 2009) (Figure 2.1). The core subsystems are (a) the resource system (e.g., a fishery), (b) the resource units within or produced by the resource system (e.g., fish), (c) the people who use and rely on the resource system and resource units (e.g., fishers and local communities), and (d) the governance system designed to manage or regulate how resource users interact with resources (e.g., through government agencies, regulatory authorities, and/or non-governmental organisations: NGOs) (Anderies et al., 2004; Ostrom, 2007a, 2009). Each subsystem influences or impacts other subsystems and comprises multiple second-tier variables; interaction outcomes also feed back into each subsystem and influence system operation (Ostrom, 2007a, 2009). SESs are also embedded within a broader social, economic, and political context, and may interlink with related ecosystems (Berkes, 2003; Ostrom, 2009).

Ostrom's framework highlights the dynamic, complex, non-linear, and evolutionary character of SESs (Holling et al., 1998; Ostrom, 2007a), and provides a practical conceptual foundation for (i) investigating system operation and sustainability, (ii) understanding relationships between variables that mediate subsystem interactions, (iii) meshing governance arrangements with specific objectives, and (iv) planning conservation actions (Anderies et al., 2004; Ostrom, 2009). Although the ecological variables within a SES are likely to be diverse, the social variables may be more extensive (Berkes, 2004; Ostrom, 2007b), which suggests that an interdisciplinary perspective will be most productive for management purposes (Heemskerk et al., 2003; Reyers et al., 2010).

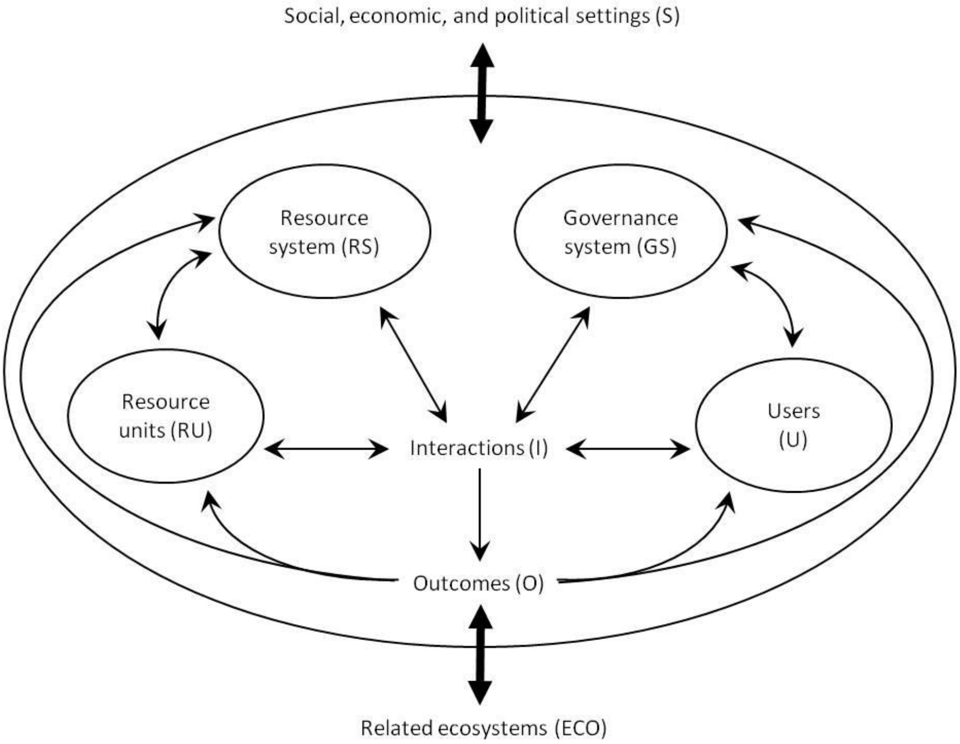


Figure 2.1. Core subsystems in a framework for analysing social-ecological systems (Ostrom, 2009). Subsystems interact with each other to produce outcomes, and these outcomes feed back to influence the operation of each subsystem and its component variables.

2.1.2 The need for social dimensions in conservation planning and management

Setting aside areas to preserve biodiversity, key species, and ecosystem services has become common conservation practice. Margules and Pressey (2000) suggest that the degree to which these areas are successful depends largely on the planning process undertaken. Although terminology or structure may vary, the fundamental principles of conservation planning continue to be relatively consistent. Conservation planning involves "identifying spatially explicit priorities and actions for the conservation of a region's biodiversity" (Reyers et al., 2010 p.958) and is considered essential for effective conservation management (Smith et al., 2006; Margules and Sarkar, 2007). However, many conservation projects are unclear about what they are trying to accomplish, how to accomplish it, and whether or not it has been successful (Sutherland, 2000); as a result, it is often difficult to convert conservation plans into action (Balmford and Cowling, 2006; Knight et al., 2006b; Reyers et al., 2010). Unachievable plans often arise from setting (overly) ambitious conservation targets, while setting targets too low leads to plans that are inadequate for protecting biodiversity (Carwardine et al., 2009). Adverse impacts on resource users and other stakeholders are unavoidable when conservation and social goals are poorly defined (Adams et al., 2011).

A broad range of approaches, research methods, software, tools, and simulation techniques have been developed over recent decades to improve the efficacy of conservation planning (Reyers et al., 2010), often attempting to promote a common framework for conceptualising the key systems, processes, or interactions involved (Rademeyer et al., 2007; Ostrom, 2009; Pressey and Bottrill, 2009). The evolution of conservation planning demonstrates that a multi-disciplinary approach incorporating stakeholder participation not only facilitates better decisions and outcomes but also adaptive management over the long-term (Heemskerk et al., 2003; Berkes, 2004).

Historically, the core focus of conservation planning has been to create and manage networks of protected areas or reserves (Smith et al., 2006). However, expansion of protected areas worldwide, and predominantly in Africa, has spawned considerable debate about their efficacy for conserving wildlife populations (Schwartzman et al., 2000; Johannesen, 2007). For example, unsustainable wildlife harvesting for human consumption continues within protected areas in many regions, threatening wildlife populations as well as the livelihoods of people who depend on wildlife resources (Garcia and Goodman, 2003; Robinson and Bennett, 2004). Similarly, the traditional ‘fences & fines’ approach to conservation management excludes local people and is widely perceived to have failed in developing countries (Barrett and Arcese, 1995; Songorwa, 1999; Holmes, 2003). In response, calls for participatory approaches have strengthened (Adams, 1998; Berkes, 2004), so that involving local communities in conservation activities has become mainstream practice (Wells et al., 1992; Western and Wright, 1994; Alpert, 1996; Inamdar et al., 1999; Hulme and Murphree, 2001).

Community-based approaches to conservation emphasise the importance of human dimensions in understanding ecosystems and managing resources (see Hulme and Murphree, 2001), and recognise the value of including local stakeholders throughout the planning process (Pierce et al., 2005; Reed, 2008). Cowling (2005) notes that successful conservation depends primarily on understanding the people involved and the choices they make rather than biological systems. While it is critical to understand biophysical processes to address environmental and resource-use problems (Anderies et al., 2004), this understanding needs to be integrated with knowledge of the drivers of human activity impacting biological systems (Redman, 1999). Accordingly, to manage ecosystems effectively it is necessary to focus on the role of humans within these systems, the influence of humans in shaping system dynamics, and interactions between human and biological factors (Berkes, 2003; Heemskerk et al., 2003; Berkes, 2004).

Sustainable resource use should be a goal of conservation planning and involve more than considering or managing each resource independently of others (NRCS, 2009). The meaning assigned to ‘sustainability’ in this thesis is that resources are used in a manner that balances their production and human harvesting, and ensures future resource production and ecosystem function are not compromised; this is consistent with the widely-used definition provided in the United Nations Brundtland Report (World Commission on Environment and Development, 1987). Striking a balance between (a) protecting the ecological processes that sustain natural resources and (b) the economics and social needs of local stakeholders should allow stakeholders to work together to achieve common goals while also ensuring benefits for future generations. However, despite the growing body of literature focusing on how to best use limited funding for conservation (Moore et al., 2004; Naidoo and Adamowicz, 2006; Naidoo et al., 2006; Murdoch et al., 2007), there has been very little research attention to the costs and benefits of conservation for local people, or the behaviour of resource users (Balmford and Whitten, 2003; Hockley et al., 2005; Balmford and Cowling, 2006).

2.1.3 Local people and natural resource management

Stakeholders are those who rely on, have an interest in, and/or influence the use of natural resources, and typically include local people (such as fishers, farmers, hunters, and pastoralists), non-governmental organisations (NGOs), government agencies, and commercial associations or industry (Cowling, 2005; Margules and Sarkar, 2007). Conservation strategies are more likely to be accepted and implemented if (i) the resource users most affected by conservation actions are involved from the outset, (ii) the strategy is practical, affordable, and achievable, (iii) techniques to monitor and review progress are included, (iv) the tools developed are easy to use, and (v) the courses of action involved are transparent to all stakeholders (Sutherland, 2000; Pierce et al., 2005; Knight et al., 2006b; Margules and Sarkar, 2007). Although it is widely recognised that all stakeholders should be identified and involved throughout the planning process, there is often little evidence of such participatory planning (Margules and Pressey, 2000; Cowling and Pressey, 2003; Beger et al., 2004).

The advent of community-based approaches to conservation has highlighted the need to incorporate costs and trade-offs for local people when implementing strategies to manage natural resources (Emerton, 1999; Berkes, 2004; Naidoo et al., 2006). For example, many studies of human-wildlife conflict assess the direct costs (e.g., crop or livestock losses) as well as the indirect costs (e.g., increased time guarding crops or livestock) of conflict to local people to understand their responses to problems with wildlife (Hill et al., 2002; CARE et al., 2003; Woodroffe et al., 2005). Johannesen (2007) presents a bio-economic model demonstrating that the expansion of protected areas could reduce the level of effective conservation and negatively impact the livelihoods of local people if the economic interests of those people are not taken into account. In many instances trade-offs for conservation are opportunity costs where local people forego alternative livelihood activities, particularly because they may have limited capacity to engage in other income-generating pursuits (Hackel, 1999; Naidoo et al., 2006; Milner-Gulland and Rowcliffe, 2007). Indirect costs can often be estimated by an increase in effort, such as time or distance travelled, for a person to use a resource. The long-term outcomes of management strategies may be undermined if local people perceive that they are subsidising conservation agendas (Hackel, 1999; Hockley et al., 2005).

Resource-user decisions and behaviour can be influenced by many social, economic, and/or political opportunities, incentives, and constraints, as well as cultural norms and value systems that mediate the legitimacy of particular courses of action (Berkes, 2003; Reyers et al., 2010). Accordingly, factors such as traditions, history of resource use, dependence on an ecological system, knowledge about that system, and access to tools and technology can determine how resource users behave within a social-ecological system (Ostrom, 2007b). The extent to which resource users invest in their livelihood (e.g., in terms of time and equipment) may also influence behaviour; itinerant users and those with low levels of investment may have less commitment to resource sustainability and management initiatives (McClanahan et al., 2008b). The inclusion of even simplified assumptions about the behaviour of resource users can dramatically affect model output and therefore understanding of the dynamics of the system being modelled (Smith and Wilen, 2003; Milner-Gulland and Rowcliffe, 2007). It can also be informative to understand how local people allocate time because this can influence their capacity to adapt, bear costs, or access conservation benefits (Colfer et al., 1999; Muller and Albers, 2004).

Despite increasing recognition that it is essential to understand fisher behaviour for effective fisheries management (Hilborn, 2007), few empirical studies have focused on small-scale fisheries (Salas and Gaertner, 2004; Abernethy et al., 2007) and even fewer examine artisanal, multi-species freshwater fisheries (Béné and Tewfik, 2001; Begossi, 2006). The spatial distribution of fishing effort can impact trends in catch rates and management plans to promote sustainable harvests (Abernethy et al., 2007; Daw, 2008). Data regarding fishers' spatial behaviour can directly inform planning and design of spatially-explicit management tools such as no-take zones (Daw, 2008). Understanding the distribution of fishing effort requires knowledge of spatial fishing patterns and fishers' decisions about how effort is allocated (Béné and Tewfik, 2001).

While fishery managers have generally made simplistic and aggregated assumptions about fisher behaviour, empirical studies have demonstrated that fishers are rarely homogeneous in their behaviour (Béné and Tewfik, 2001; Salas and Gaertner, 2004). Based on the assumptions that fishers have knowledge of resource distribution and unconstrained access to resources, the ideal free distribution (IFD) predicts that (a) resource users will organise themselves to gain an equal return from harvesting and (b) harvesting pressure will increase with resource availability (Kacelnik et al., 1992; Gillis et al., 1993). Exploring selection of fishing sites by artisanal fishers Abernethy et al. (2007) found that, contrary to the predictions of the IFD, fishers (i) did not distribute themselves to ensure equal average returns for all individuals, (ii) did not have ideal knowledge of the resource, (iii) did not always seek to maximise profit, and (iv) were constrained in their choice of fishing site (also see van Oostenbrugge et al., 2001). There is increasing evidence that fisher behaviour may be influenced or constrained by many variables (see Seixas and Begossi, 1998; Béné and Tewfik, 2001; Salas and Gaertner, 2004; Begossi, 2006; Abernethy et al., 2007; Daw, 2008; McClanahan et al., 2008b), including:

- Biological and environmental factors such as stock availability, weather, and seasonal variation;
- Social and historical factors such as local rules, social norms, customary tenure and territories, gear theft, and traditional community fishing boundaries;
- Economic factors such as fishing revenues and costs, catch expectations, market conditions, and level of competition;
- Governance factors such as management regulations, restrictions on access to fishing areas, and harvest limits implemented at community level; and
- Personal factors such as fisher skills, capability, experience, perceptions, goals, preferences (gear, locations, and fish species), familiarity with fishing locations, knowledge of stock, level of acceptable risk, and time available for fishing compared to other livelihood or social activities.

This array of potential factors, particularly the range of personal variables, confirms that economic motives alone are insufficient to explain fisher spatial behaviour (Béné and Tewfik, 2001). Understanding the factors influencing spatial effort distribution will improve prediction of how spatial management interventions are likely to create or increase costs for resource users, as well as how local people are likely to respond (Abernethy et al., 2007; Daw, 2008). This understanding will promote partnership between resource users and managers, thereby enhancing governance systems (Salas and Gaertner, 2004).

2.1.4 Fisheries and fishery management

Availability and accuracy of data for natural resource management varies between ecosystems and also geographic regions (Iachetti, 2007). Most aquatic research has focused on marine systems, and the trend of decline in the average size and trophic level of species caught in marine fisheries during the last 50 years suggests that many of the world's fisheries are unsustainable over the long term (see Ludwig et al., 1993; Gracia, 1996; Johannes, 1998; Pauly et al., 1998). Inland fisheries are not immune to this effect and the resources of most large inland fisheries are currently fully utilised or overexploited (Mölsä et al., 1999; Balirwa et al., 2003; Hilborn et al., 2003; Matsuishi et al., 2006; Lorenzen, 2008; FAO, 2010; Welcomme et al., 2010). Inland fisheries may also be interlinked with wetland habitat and larger ecosystems that are biologically diverse and/or have considerable conservation value (Pidgeon, 1996; Welcomme, 2001).

2.1.4.1 Inland fisheries

Inland fisheries occur in freshwater or estuaries and include any activities conducted to extract fish and/or other aquatic organisms from inland waters (Welcomme, 2001). Capture fisheries remove aquatic organisms from natural or enhanced waters, while culture-based fisheries are maintained by stocking from aquaculture systems (FAO, 1997, 1999). Enhanced fisheries, which can include culture-based fisheries, involve "activities aimed at supplementing or sustaining the recruitment of one or more aquatic organisms and raising the total production or the production of selected elements of a fishery beyond a level which is sustainable by natural processes" (FAO, 1997, p. 4).

Fishing is an important activity for many human communities in inland rural areas (Berkes, 2003; Pomeroy and Rivera-Guieb, 2005). Inland fisheries are often essential for food security and a key source of animal protein for many people (SEAFDEC, 2005; Welcomme, 2011b). In 1997 inland fisheries accounted for 8.1% of the total world fish capture (FAO, 1999), by 2001 they accounted for 9.3% (FAO, 2003), and by 2009 they accounted for 10.3% of the capture (Welcomme, 2011b). Almost 90% of this catch occurs in developing countries (FAO, 2003; SEAFDEC, 2005) and most of the countries with major inland fisheries are in Asia and Africa (Welcomme, 2011b). Although catch data reported to the Food and Agricultural Organization of the United Nations (FAO) indicates steady increases in production from inland capture fisheries, this could be linked to the number of fisheries reporting catches; information about actual catch sizes and levels of consumption for specific fisheries is very limited (SEAFDEC, 2005). Because of the dispersed and informal nature of many inland fisheries there is considerable underreporting of catches and fishing activity (Pauly et al., 1998; Caddy and Garibaldi, 2000; Allan et al., 2005). As a result, inland catches and their role in providing food security and protein for rural communities are often considerably underestimated, and are probably greater than the amount reported to FAO by a factor of 2 or 3 (FAO, 1999).

2.1.4.2 Small-scale inland fisheries

Small-scale inland fisheries in tropical regions are usually multi-species and multi-gear fisheries (SEAFDEC, 2005). Reliable data for these fisheries are rare; data that are available are often inadequate to show broad-scale trends (Pauly et al., 1998) and local depletions of fish stocks may not become apparent until it is too late (Sodhi et al., 2007). Data deficiencies are also evident in developed countries, understating the socioeconomic value of inland fisheries worldwide (Berkes, 1990). The need for an improved understanding of small-scale fisheries is gradually being recognised (Hauck, 2000), and an increasing number of studies are focusing on these fisheries in both marine (Johannes, 1998; Hawkins and Roberts, 2004) and freshwater ecosystems (Sodhi et al., 2007). In many cases management decisions for small-scale fisheries are made with insufficient knowledge of catch sizes, fishing effort, or length-weight relationships (Welcomme, 2001). Whereas conventional conservation methods and resource surveys often fail to account for the dynamics of these fisheries, information from local fishers can be invaluable for conservation planning and management by incorporating a greater understanding of traditions, livelihoods, subsistence needs, and costs to fishers (Cinner et al., 2008).

2.1.4.3 Fishery management

Historically, fisheries management has been conducted using a 'top-down' approach whereby regulatory agencies develop and impose rules and restrictions on fishers to control fishing activity (Berkes, 2003). Fishery management has also generally been reactive rather than proactive, had a single-species and biological focus, and struggled to implement effective systems of governance for smaller-scale systems (Berkes et al., 2001; Berkes, 2003; Mapstone et al., 2008). Over the past decade fisheries science has increasingly adopted an ecosystem approach to management (Christensen and Pauly, 2004). This shift has coincided with expectations that fisheries be managed on a participatory basis with resource users and other stakeholders, and take into account broader-scale issues such as socioeconomic sustainability, maintaining ecosystem biodiversity and function, and conservation objectives (Sainsbury et al., 2000; Berkes, 2003; Mapstone et al., 2008; Levin et al., 2009). In small-scale fisheries there is often considerable uncertainty regarding fisher behaviour and the costs to fishers of management actions (Holland and Herrera, 2009; Daw et al., 2011b; Wise et al., 2012).

In general, the first step in conservation management is to identify broad objectives that include long-term interests and avoidance of irreparable damage. In the fisheries realm, these objectives typically include (i) catches being as large as possible with a low probability of substantial stock depletion, (ii) maintaining catches at relatively stable levels, and (iii) limiting the environmental impacts of fishing to acceptable levels (FAO, 1995). Fishery management options that can be used to achieve these objectives include temporal and/or spatial closures, gear restrictions, minimum and/or maximum limits for the size of fish that can be caught, and fishery enhancements (Welcomme, 2001; SEAFDEC, 2005). Because extensive fishing of one or more species may enhance biodiversity, it is important that management options consider fishery sustainability in conjunction with biodiversity conservation (Balirwa et al., 2003).

Temporal closures or closed seasons involve restricting all methods of fishing for a period of time each year to protect juvenile fish or the spawning stock (i.e., fish that are mature and breeding) from fishing mortality (Martin-Smith et al., 2004; Musick and Bonfil, 2005; SEAFDEC, 2005). The respite from fishing pressure is intended to be of sufficient duration to allow rebuilding of the stock (FAO, 1995) and a thorough understanding of the biology of each species involved is required to ensure that the timing of the closure is effective (Sutinen, 1999). Closed seasons can be complicated to implement, particularly for a multi-species fishery, and are frequently insufficient to prevent fishery collapse without other measures (Sutinen, 1999).

Spatial closures or no-take zones involve setting aside regions where fishing activity is excluded and are designed to provide refuges for fish (mainly the spawning stock) while protecting a variety of fish habitats (Johannes, 1978; FAO, 1995; Johannes, 1998; Roberts et al., 2001; Sale et al., 2005). The primary objective of spatial closures is to maintain essential ecological processes for the fishery (Cochrane, 2002) and thereby improve fishery sustainability (Béné et al., 2007). Debate regarding the effectiveness of protected areas for conserving wildlife populations (see Tupper et al., 2002; Halpern, 2003; Johannesen, 2007) is especially prevalent in relation to marine fisheries where marine protected areas (MPAs) have been implemented to increase yields for adjacent harvested areas (Roberts et al., 2001; Pauly et al., 2002; Gell and Roberts, 2003; Hilborn et al., 2004; Béné et al., 2007). No-take zones may be most effective when combined with gear restrictions in adjacent areas (McClanahan et al., 2008b). Although designation of these areas has often been based on key ecological criteria, factors such as the economic value and social importance of the resource for local people have largely been ignored (Smith and Wilen, 2003; Pollnac et al., 2010). These oversights are likely to compromise the effectiveness of the protected areas over time (Beger et al., 2004).

Use of more-efficient and habitat-destructive fishing gear has increased markedly throughout the tropics in recent years and is probably the greatest threat to the long term viability of fish populations (Sodhi et al., 2007). Gear restrictions generally prohibit use of any fishing methods that are directly damaging to habitat (such as dynamite or poison) and those that take a disproportionate amount of juvenile fish (Welcomme, 2001). Limitations on minimum mesh size are common and intended to protect breeding stocks of fish species (Welcomme, 2001). However, this method of regulation can be problematic within multi-species and multi-gear fisheries where a single mesh size limitation is often inappropriate (Welcomme, 2001).

Fishery managers also frequently restrict the size of fish that can legally be caught; minimum fish lengths are typically species-specific and are intended to protect the breeding stocks (Welcomme, 2001). Although these limitations may be difficult to monitor in traditional artisanal fisheries where there are many points of sale for catches, some enforcement is feasible by prohibiting commercial traders from buying fish below the minimum size (Welcomme, 2001). Maximum size limits can also be implemented, and benefits often depend on the relationship between species-specific fecundity and body size. Because larger fish are generally more fecund than smaller fish, protecting large female fish will usually lead to greater recruitment (Blueweiss et al., 1978; Roberts et al., 2001; Halpern, 2003).

Although stock enhancements are designed to enhance, conserve, or restore fisheries, they are of limited use for heavily exploited stocks (Lorenzen, 2008). Furthermore, hatchery releases are costly and the economic feasibility of continued enhancements needs to be considered, particularly in developing countries where financial resources are limited (Lorenzen, 2005). Enhancements are at best a temporary solution only to be considered as secondary measures in conjunction with other management options (Lorenzen, 2005).

Compliance with any of the management strategies outlined above is likely to be problematic within small-scale artisanal fisheries if local people are not considered to be key stakeholders and consulted throughout all stages of the management process. Levels of resentment and non-compliance are often high when management strategies are imposed on local people (Milner-Gulland and Rowcliffe, 2007). Because fisheries management has become increasingly concerned about the livelihoods of people who rely on fishery resources (Berkes, 2003; Welcomme, 2003), many managers recognise that understanding stakeholder behaviour may be more important for a fishery's long-term sustainability than understanding the ecology of fish species present (Béné and Tewfik, 2001; Salas and Gaertner, 2004; Hilborn, 2007). Accordingly, an improved understanding of livelihood impacts within small-scale fisheries is critical to inform the development of appropriate management policies (Allison and Ellis, 2001; Smith et al., 2005; Cinner et al., 2011).

2.1.5 Governance and management

Because an array of factors may influence the decisions of resource users within a dynamic social-ecological system, it is frequently difficult to manage resource user behaviour in ways that are consistent with conservation goals (Berkes, 2003). An effective and inclusive system of governance is integral to successful management over the long term, particularly for open-access resources (Dietz et al., 2003). A governance system is typically a multiple-level arrangement of institutions and rules to oversee conduct within a social-ecological system (Ostrom, 2007b). A governance structure that is effective in one case might not be suitable in others and may need to be adjusted over time (Brondizio et al., 2009; Marine Resources Assessment Group, 2009). The governance system needs to operate at appropriate scales to reflect the characteristics of the resource system as well as the attributes of the resource users; governance practices at local level may have greatest impact on the effectiveness of management strategies (Berkes, 2003; Ostrom, 2007b).

The 'Tragedy of the Commons' (Hardin, 1968) refers to circumstances where open-access resources continue to be exploited by many individuals even though this may deplete stocks and decrease harvests considerably over time and eventually lead to resource extinction. The reasoning or justification for this is that any restraint in exploitation by one individual increases the opportunity for others to exploit the resource and gain a disproportionate share. It is widely assumed that this process occurs wherever there is competition between individuals for resources that are not privately owned and is inevitable without government regulation (Sutherland, 2000; McClanahan et al., 2008b; but see Ostrom, 2007a). While some advocate that fishing communities in many regions often successfully avoid overexploitation of their resource through informal self-management (see Leal, 1998), others support a more formal management arrangement where planning and regulations are established with community involvement (Pomeroy, 1995; Wiber et al., 2004; Cardoso et al., 2005; Pitt, 2007).

The benefits of incorporating resource users into the governance system for a SES is that policies and rules are more likely to be founded on local knowledge, better understood, locally appropriate, broadly accepted, and require reduced enforcement costs (Anderies et al., 2004; Ostrom, 2007b, 2009). However, if the governance system is too highly decentralised it may lack leadership or be dominated by people with conflicting agendas, lack independent external mechanisms for resolving conflict between resource users, and/or be unable to cope with large-scale management issues (Ostrom, 2007b, 2009). Participation by resource users when developing policies and rules will also make the governance system more transparent and adaptable to varying circumstances (Berkes, 2003; Pomeroy and Rivera-Guieb, 2005; Ostrom, 2007b), and increase the likelihood that resource users will share and communicate management goals (Reyers et al., 2007) and enter into co-management arrangements (Jentoft et al., 1998; Cinner et al., 2012).

Compliance with rules and regulations for natural resource management is often more a function of the processes and partnerships involved in establishing a governance system than purely level of enforcement (Pollnac et al., 2010). However, compliance might also depend on how boundaries for resource use are defined (Anderies et al., 2004), issues of customary tenure (Foale and Macintyre, 2000; Cinner, 2005; Mills et al., 2010), and the adaptive capacity of resource users (i.e., those with low adaptive capacity will be less able to comply irrespective of the extent to which they support management objectives) (McClanahan et al., 2008a). Non-compliance with rules and regulations can compromise the functioning of SESs, and typically involves short-term as well as long-term loss of revenue for legitimate resource users (Gavin et al., 2010; McCook et al., 2010). Interactions between resource users and institutional entities that oversee governance practices have rarely received research attention when examining SES function, overlooking the role that resource users can play in effective governance (Anderies et al., 2004).

2.1.6 Adaptive management

Adaptive management is a structured process of ‘learning by doing’ (Walters, 1997) and comprises implementing management strategies as experiments to be tested, learning from the outcomes of these actions, and using the information derived from this process to adjust strategies (Lee, 1999; Milner-Gulland and Rowcliffe, 2007). Adaptive management acknowledges that resource managers often have incomplete knowledge of ecosystems but can increase understanding of systems dynamics over time through feedback learning (Lee, 1999; Berkes, 2003), similar to development of traditional ecological knowledge (TEK) by resource users (Berkes et al., 2000). An adaptive management plan includes (i) a management policy specifying actions based on existing information about a system, (ii) a plan for monitoring system responses to management actions, and (iii) a system to implement a management policy (Parma and NCEAS Working Group on Population Management, 1998). Trade-offs between anticipated costs and benefits of alternative management policies are often evaluated prior to implementation to determine which policy is most likely to meet objectives (Parma and NCEAS Working Group on Population Management, 1998). An adaptive approach has been used to manage coastal marine ecosystems (Walters, 1997), river systems (Walters, 1997), fisheries (Berkes, 2003), and forests (Lee, 1999).

Because adaptive management addresses uncertainty and is inherently flexible, it does not postpone management action until knowledge of system dynamics is complete (Lee, 1999; Brown et al., 2012). Modelling techniques are frequently used to narrow choices about which management options are likely to be effective. Although these techniques often reveal gaps in knowledge, a tendency to over-model may slow the adaptive management process (Walters, 1997). Also, where adaptive management involves considerable experimentation with insufficient information, it may involve risks for species that are already well-adapted to current conditions or for ecosystems more broadly (Walters, 1997; Parma and NCEAS Working Group on Population Management, 1998).

The adaptive management process can be participatory to promote involvement and collaboration between stakeholders (Berkes, 2003); it is also relatively transparent, revealing the efficacy of management actions and who they benefit in addition to ecosystem responses (Lee, 1999). Adaptive management can be particularly effective when it draws upon local fishers' knowledge to manage small-scale fisheries in developing countries that often lack scientific information (Lee, 1999; Berkes et al., 2000); it also highlights the importance of monitoring to achieve objectives, because failing to monitor precludes opportunities to learn (Parma and NCEAS Working Group on Population Management, 1998).

2.1.7 Monitoring

Monitoring is critical for effective management of social-ecological systems (Kremen et al., 1998; Gavin et al., 2010) and involves measuring and assessing a defined range of variables to determine patterns and changes over time (Yoccoz et al., 2001; Hauser et al., 2006). Monitoring should occur for each component subsystem of a SES and generate information about the state of the resource system, the behaviour of resource users, and interactions between users and resource units (Welcomme, 2001; Branch et al., 2006). For a fishery this would include measuring fisher behaviour and fishing effort, catch sizes and composition, levels of compliance, and ecological changes (Branch et al., 2006; Hauser et al., 2006). Ideally monitoring should be conducted using multiple methods, and particularly observational techniques, simultaneously to improve the accuracy and reliability of results through triangulation (Lunn and Dearden, 2006; Gavin et al., 2010).

Monitoring helps to gauge the success or shortcomings of strategies and ensure objectives are being achieved (Sutherland, 2000); results can be used to adjust conservation actions over time via adaptive management (Kremen et al., 1993; Sutherland, 2000; NRCS, 2009). Because monitoring provides feedback about system processes and operation it directly facilitates adaptive management (Armitage, 2003; Uychiaoco et al., 2005; Milner-Gulland and Rowcliffe, 2007) while increasing the probability of compliance with rules and regulations (Abbot and Guijt, 1998; Anderies et al., 2004; Ostrom, 2007b). However, monitoring is frequently expensive and some argue that the cost of monitoring should be evaluated in terms of benefit(s) to decision-making (Hockley et al., 2005; Hauser et al., 2006).

The costs of monitoring can often be reduced by adopting a participatory approach and involving resource users (Abbot and Guijt, 1998; Milner-Gulland and Rowcliffe, 2007). At broadest scale this could include drawing upon traditional ecological knowledge held by local people about the resource system and its dynamics (Berkes et al., 2000; Berkes, 2004). For example, fishers usually have detailed knowledge of the local area, fishing patterns, and gear use (including harmful methods), as well as the characteristics and habitat requirements of fish; fishers are also frequently aware of factors that may affect the health of fish or functioning of the fishery (FAO, 1995; Welcomme, 2001). This information can be used to ensure monitoring protocols have relevance for local people (Berkes, 2003).

Locally-based or participatory monitoring is carried out at community scale by people who do not have prior formal training in monitoring techniques; resource users are directly involved in data collection and, where appropriate, analysis (Danielsen et al., 2005; Halls et al., 2005). Danielsen et al. (2009) suggest a typology of locally-based monitoring techniques categorised by the degree of participation by local people compared to involvement by professional scientists; each category has strengths and weaknesses, and the most relevant monitoring system should be selected on a case-by-case basis.

Participatory monitoring can engage local communities in resource management while simultaneously collecting key information for conservation planning (Ticheler et al., 1998; Welcomme, 2001; Milner-Gulland and Rowcliffe, 2007). Integrating locally-based monitoring with planning and management can improve participants' confidence in results and also make managers more accountable to resource users (Pound et al., 2003). However, Uychiaoco et al. (2005), Lunn and Dearden (2006), and Danielsen et al. (2009) suggest that the accuracy of information collected by local resource users may be uncertain or less precise than data collected by professionals. On the other hand, relying only on information from professionals might discount crucial local knowledge (Rist et al., 2010). Although participatory monitoring may be less costly for managers than traditional scientific surveys, it is often the case that local people bear the costs of monitoring, and without a substantive benefit for local people this form of participation will not be sustainable (Hockley et al., 2005). Combining participatory and science-driven approaches to monitoring may derive more accurate and relevant indicators of conservation progress than either approach could achieve independently (Lewis and Phiri, 1998). Most importantly, working in partnership with resource users and other local stakeholders can also promote a greater sense of ownership and responsibility for managing the resources they rely on, as well as a commitment to finding solutions to problems that arise (Neiland et al., 2005; Fraser et al., 2006; Sitati and Walpole, 2006).

2.1.8 Systematic conservation planning

Systematic conservation planning has developed rapidly over the past decade as a framework for identifying priority areas for conservation action, based on explicit goals and/or constraints (Margules and Pressey, 2000; Margules and Sarkar, 2007). Although originally focussing on terrestrial habitats, the approach is increasingly used when designing marine protected areas (Ball and Possingham, 2000; Stewart and Possingham, 2005; Ban, 2008; Maiorano et al., 2009) and to incorporate ecosystem services into the planning framework (Chan et al., 2006; Cowling et al., 2008). A number of software tools have been developed to inform and assist the systematic planning process, including Marxan (Ball and Possingham, 2000) with the CLUZ (Conservation Land-Use Zoning) (Smith, 2004) and MinPatch (Smith et al., 2010) extensions, Marxan with Zones (Watts et al., 2009), CREDOS (Conservation Reserve Evaluation & Design Optimisation System) (Crossman et al., 2007), and C-Plan (Pressey et al., 2005). These software packages have been widely used to allow decision-makers to model and explore the potential implications of conservation and management strategies under a range of scenarios (Cowling and Pressey, 2003; Sarkar et al., 2006; Iachetti, 2007; Game et al., 2009); however, costs are often determined from a relatively narrow perspective and might not incorporate interactions between multiple costs at a site (see Stewart and Possingham, 2005).

Naidoo et al. (2006) identify five types of conservation cost: acquisition costs (for property rights), management costs (establishing and maintaining a conservation program), transaction costs (negotiating a conservation program with stakeholders), damage costs (loss of economic activities due to a conservation program), and opportunity costs (foregone opportunities). Local people typically bear damage and opportunity costs (Balmford and Whitten, 2003). Although conservation costs are spatially variable (Naidoo et al., 2006), they are rarely specified at fine scale in conservation planning, and surrogates are frequently used (Ban and Klein, 2009). Surrogates often involve uniform costs that do not reflect the actual costs of conservation for stakeholders in complex SESs (Naidoo et al., 2006; Adams et al.,

2010; Weeks et al., 2010c). For example, plans that do not take into account local tenure systems and/or the relatively low spatial mobility of subsistence or artisanal resource users, may misrepresent costs considerably and lack support from local people (Richardson et al., 2006; Weeks et al., 2010c; Grantham et al., 2012). This will in turn reduce compliance and increase management costs through a need for increased enforcement (Weeks et al., 2010b), compromising efforts to use funds for conservation efficiently and within budgetary constraints (Richardson et al., 2006; Weeks et al., 2010c). Accounting for spatial variation in costs can not only enable funds to be used most effectively but also minimise conflict that arises when protected areas restrict resource users excessively (Ban and Klein, 2009; Weeks et al., 2010c).
Similarly, the greater the extent to which conservation interventions, such as protected areas, are aligned with customary practices and the local socioeconomic context, the more likely it is that they will involve fewer and lower costs to resource users (Cinner, 2007). Interventions should also aim to distribute costs and benefits equitably among resource users (Cinner, 2007). Recent research emphasises the importance of incorporating spatially-explicit socioeconomic information (such as costs for resource users) into conservation planning from the outset in order to more accurately assess the potential impacts of different management scenarios (Stewart et al., 2003; Bode et al., 2008b; Cameron et al., 2008; Carwardine et al., 2008; Polasky, 2008). However, socioeconomic data (as well as ecological data) are typically lacking in developing countries, particularly at spatial scales relevant to conservation planning (Ban et al., 2009; Weeks et al., 2010c).

2.2 CONCEPTUAL FRAMEWORK FOR THE RESEARCH

This interdisciplinary study focuses on conservation planning and management within a dynamic social-ecological system. Using insights and methods from social sciences, fisheries management, and ecology, the research investigates how behavioural and socioeconomic data can be incorporated into the conservation planning process, including the nature of data to collect and how best to collect it.

At very broad scale, Figure 2.2 illustrates how resource management and conservation have been approached historically in a given social, economic, and political context.

Generally, governance systems are intended to regulate the behaviour of resource users (e.g., fishers), influencing how resource users interact with the resource system (e.g., a fishery); governance is then adjusted according to the performance of the resource system. These relationships are indicated by solid arrows in Figure 2.2. This framework does not take into account the feedback that resource users can provide about the state of the resource system (e.g., in terms of catch sizes, fish sizes, and fishing effort), the role resource users can play in the governance system, or the way resource distribution and dynamics affect the behaviour of resource users. These relationships are central to my research and often overlooked or understated in conservation planning; they are indicated by dotted arrows in Figure 2.2. The interactions shown in Figure 2.2 are consistent with Ostrom's (2009) framework for analysing the dynamics and sustainability of social-ecological systems. However, Ostrom's framework includes greater complexity, highlighting that (i) all components of SESs interact, (ii) the behaviour of resource users is integral to system operation, and (iii) information about resource users should influence the governance system directly, thereby impacting the resource system and management outcomes.

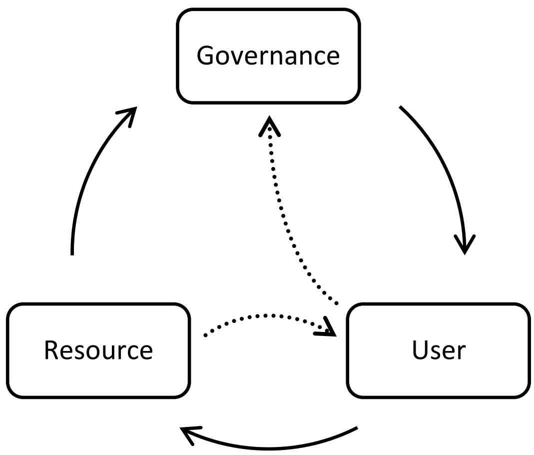


Figure 2.2. Diagram illustrating the historical approach to conservation (interactions shown by solid arrows) and links that have traditionally been lacking in conservation planning (interactions shown by dotted arrows).

The conceptual framework for my research draws upon elements from each of the approaches and tools outlined in Section 2.1 and is illustrated in Figure 2.3. This framework focuses on the interactions between resource users and mechanisms of governance that impact system management and conservation. The interactions addressed by my research are shown in bold in Figure 2.3. The case study is a large inland fishery at Lake Alaotra, Madagascar, where management and governance are directed towards sustaining two resource systems: 1) the marsh ecosystem as habitat for wildlife and an area of conservation importance (e.g., to protect the critically endangered Alaotran gentle lemur), and 2) the fishery system, which is of economic importance to fishers and plays an important role in marsh conservation (e.g., the marsh provides habitat for fish while a viable fishery will reduce pressure to convert the marsh for growing rice).

With reference to Figure 2.3 and in the context of my study, (i) the resource system is the Lake Alaotra fishery, (ii) the resource units are fish populations, (iii) the resource users are artisanal fishers, and (iv) the governance system comprises regional fishing by-laws and management actions by national and regional government agencies, the local Federation of Fishers, and Durrell Wildlife Conservation Trust. Although it is beyond the scope of this study to examine the psychological basis for resource-user behaviour, it is important to be aware that the motivations of resource users are likely to reflect factors that affect them as individuals and also as a group. Motivations may be based on short-term or immediate needs, longer-term goals, values and beliefs, and/or intrinsic factors (Cabrera and Defeo, 2001; Salas and Gaertner, 2004).

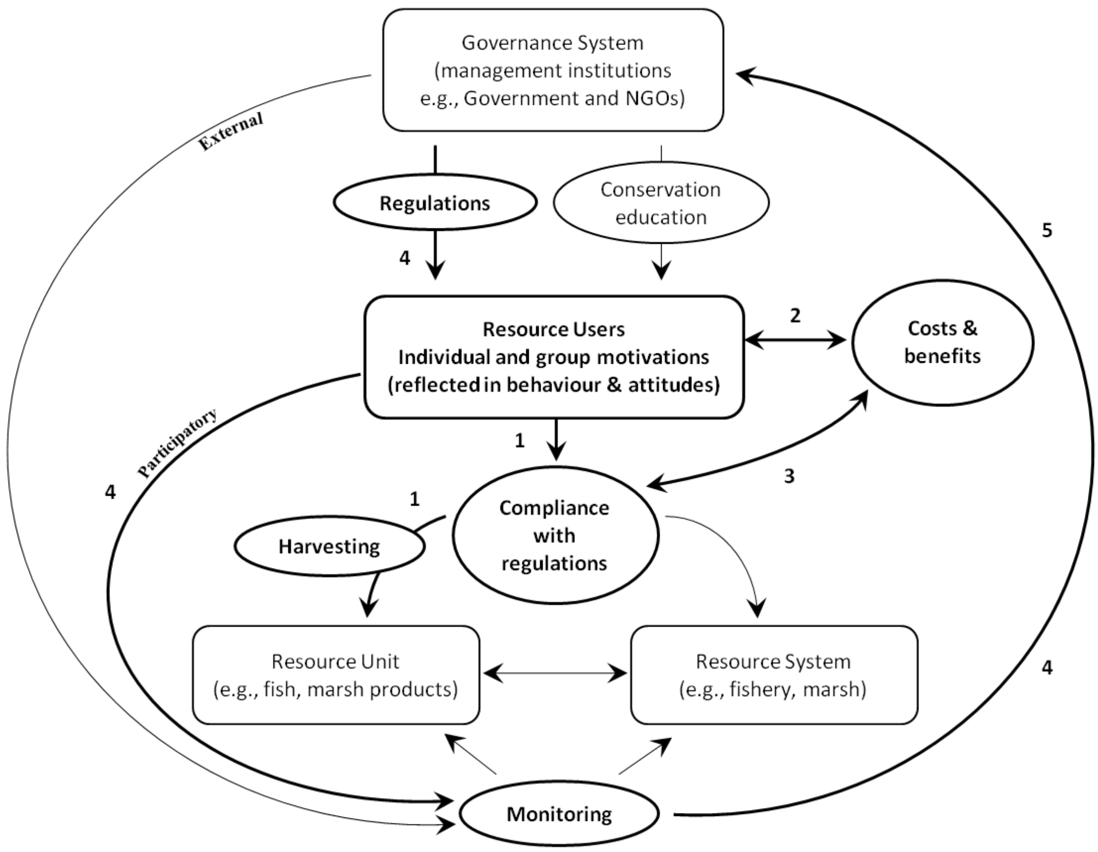


Figure 2.3. Schematic diagram illustrating the conceptual framework for the research, focusing on interactions between management institutions, regulations, and resource users. Entities are shown in boxes. Actions are shown in ovals. Components in bold are addressed by the research, using Lake Alaotra as a case study. Numbers refer to interactions where the research objectives (listed in Section 1.2 in Chapter 1) are addressed.

Institutional factors (such as regulations and/or education) relating to management and conservation actions may also influence fisher behaviour or attitudes, including compliance with the governance system. In turn, fishers' responses will impact the effectiveness of management strategies and conservation interventions. Institutional and management factors that may influence the behaviour and attitudes of fishers at Lake Alaotra include: (i) fishing regulations such as gear-based restrictions, spatial and temporal restrictions, and minimum fish lengths, (ii) regulations for use of the marsh and banning fires, (iii) governance practices, (iv) conservation education, (v) extent of community involvement in management, and (vi) the appropriateness of regulations.

The oval depicting regulations in Figure 2.3 refers to development, implementation, and enforcement of regulations. Links between resource-user behaviour, compliance with regulations, and harvesting patterns are components of characterising the Lake Alaotra fishery (objective 1 of the research). The costs and benefits that regulations involve for resource users will influence user behaviour and attitudes (objective 2), ultimately determining the extent to which users are willing and/or able to comply with regulations (objective 3). Costs and benefits may be user-specific and primarily determine compliance at the level of individual resource users. Compliance or non-compliance with regulations can have a direct impact on the resource system (for example, through sustainable practices or via habitat degradation or conversion, respectively) as well as an effect on the resource unit through harvesting effort. As a result, compliance with regulations will directly impact the efficacy of efforts to conserve resource systems. The two-way links in Figure 2.3 between resource users, costs & benefits, and compliance with regulations represent the feedback structure necessary to inform governance through adaptive management, including options for monitoring methods (objectives 2 and 3). Interactions between regulations, support by fishers, and monitoring were explored to integrate spatially-explicit cost information into conservation planning, particularly for restricted areas (objective 4).

When viewed as a whole, the cycle and interactions depicted in Figure 2.3 form an adaptive management loop that allows ongoing assessment of the appropriateness of regulations; this refers to the effectiveness of regulations for achieving management objectives, how well they are aligned with resource system characteristics, how equitably they distribute costs, whether they are consistent with local social norms and customs, and whether resource users perceive them to be legitimate (objective 5).

In a review of past and future directions for conservation biology, Balmford and Cowling (2006) describe a range of interlinked challenges that are characterised by a strong need for interdisciplinary research and the importance of documenting and understanding the benefits of natural systems for human well-being. This focus acknowledges that resource users are integral to the ecosystems they draw resources from, alongside the species of wildlife and habitats that are often assigned greater weight in conservation strategies. Although it has become widely recognised that human communities must be involved throughout planning and management in order to achieve effective conservation, a framework to incorporate socioeconomic information from resource users is lacking. Approaches to conservation have frequently failed to incorporate spatial behaviour and cost-related information from resource users into the planning process. This research will advance conservation science by demonstrating an approach for integrating spatially-explicit behavioural and socioeconomic data into conservation planning, in a real case study.

CHAPTER 3 STUDY SITE & RESEARCH METHODS

3.1 STUDY SITE

3.1.1 Geography and climate

Madagascar is the world's fourth largest island and has a population of approximately 19 million people, most of whom live in rural areas and depend on subsistence farming (rice and cattle) and fishing activities for their livelihood (FCO, 2008). The study site is Lake Alaotra, the largest lake in Madagascar and the base for the nation's most productive inland fishery (Andrianandrasana et al., 2005). Located 250km north east of the capital Antananarivo and in the Alaotra-Mangoro region (see Figure 3.1), the Alaotra wetland covers a total area of 7,225km^2, is internationally recognised as an important area for biodiversity conservation, and was declared a Ramsar site[1] in September 2003 (Ramanampamonjy et al., 2003).

In 2007, the Alaotra wetland was designated as a new protected area by the government of Madagascar (Andrianandrasana, 2009). It is a complex habitat mosaic that includes open water, marsh, reedbeds (dominated by papyrus *Cyperus madagascariensis* and reeds *Phragmites communis*), and rice fields. At an elevation of 750m above mean sea level, Lake Alaotra is 200km^2 (20,000ha) in size (40km long with width varying from 3km to 8km) and has a maximum depth of 4 metres at the end of the wet season (Moreau, 1979a; Vanden Bossche and Bernacsek, 1991; Ferry et al., 2009). The marsh surrounding the lake covers 230km^2 and approximately 1,200 km^2 of rice fields are adjacent to the lake and marsh (Andrianandrasana et al., 2005).

There are two main climatic seasons in the Alaotra region: the wet season occurs from December to April and is hot with heavy rain and rising water levels, while the dry season from May to November is cooler and water levels decrease. There is up to two metres difference in water level between the high in March and the low in November (Moreau, 1979a; Ferry et al., 2009).

[1] The Convention on Wetlands of International Importance is an international treaty for the conservation and sustainable use of wetlands. The Convention was developed at a meeting in Ramsar, Iran, in 1971 and came into force in 1975. In March 2011, the Ramsar List of Wetlands of International Importance included 1,925 sites covering an area of 1,878,689km^2 (see http://www.ramsar.org/pdf/sitelist.pdf). Lake Alaotra is one of seven sites in Madagascar. Sites are known as 'Ramsar sites'.

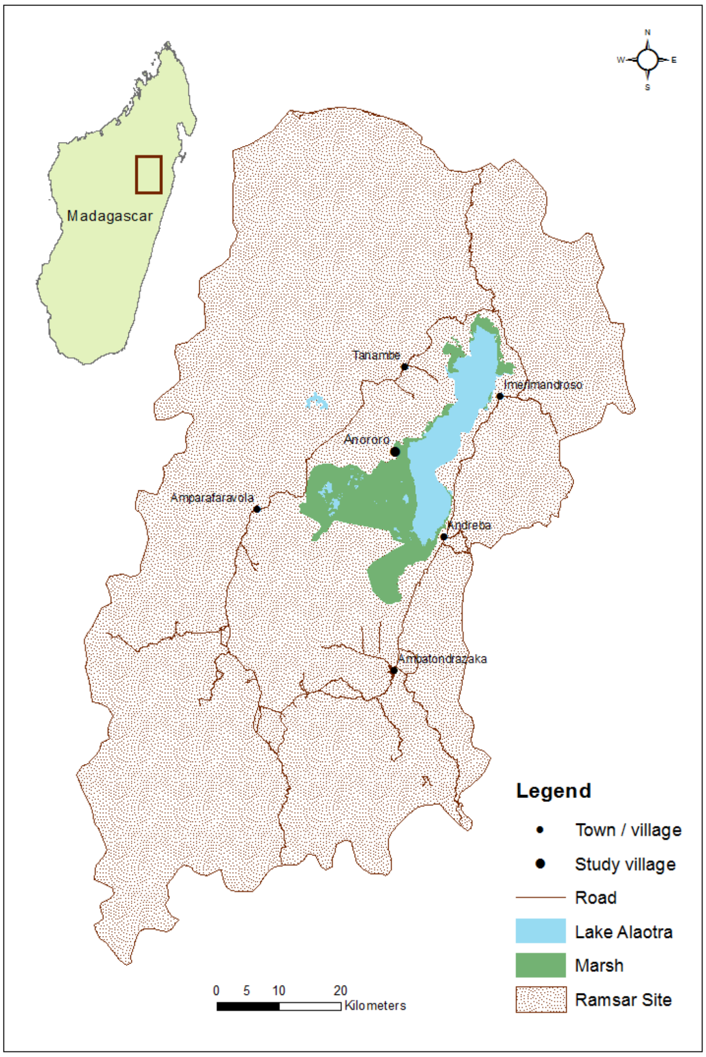


Figure 3.1. Map of Madagascar showing the location of Lake Alaotra and adjacent wetlands as well as the extent of the Ramsar Site.

3.1.2 Fish species and introductions

Since the early 1900s many exotic fish species have been introduced to Lake Alaotra in an effort to improve fishery production and provide food sources for local people (Moreau, 1979b). Before introductions began, fish native to the lake included *Paratilapia polleni* (Cichlidae – dominant fish population), *Rheocles alaotrensis* (Bedotiidae – Madagascar rainbowfishes), *Eleotris legendrei* or *Ratsirakia legendrei* (Eleotridae), *Anguilla mossambica* (Anguillidae – African longfin eel), and *Anguilla marmorata* (Anguillidae – Giant mottled eel) (Moreau, 1979b). The composition and commercial focus of Lake Alaotra's fishery has varied in line with successive fish introductions. Prior to 1955 (when *Tilapia rendalli* was introduced), *Cyprinus carpio*

carpio (introduced in 1926) was the dominant species, but by 1958 *T. rendalli* comprised the majority of fish catches (Moreau, 1979b). Since the 1960s there have been a series of introductions, each resulting in an eventual change of dominant species (Kiener, 1962; Moreau, 1979b). One of the more damaging introductions was the eastern mosquitofish (*Gambusia holbrooki*) in 1940, intended to control mosquitoes and reduce the spread of malaria (Mutschler, 2003); however, it preyed on fish larvae instead of mosquito larvae and contributed to devastation of native fish populations (Reinthal and Stiassny, 1991). Blotched snakehead (*Channa maculata*) were introduced to Madagascar in 1976 after then-President Ratsiraka observed the species at an aquaculture facility in North Korea. Snakehead is a voracious predator and has infested waterways across the country (Courtenay et al., 2004); it was first detected in Lake Alaotra in 1980 (Mutschler, 2003) and was considered established by 1985 (Courtenay et al., 2004). A range of species of tilapia, including hybrids but mainly Nile tilapia (*Oreochromis niloticus niloticus*), predominate in current catches. The lake was stocked annually with juvenile carp and tilapia from 2005 to 2007; however, stock enhancement has not occurred since due to a lack of funds as well as extensive fish mortality during previous releases (J. Rabemazava, Service Régional de la Pêche et des Ressources Halieutiques, pers. comm.).

3.1.3 Fishers and annual catches

The human population in the Lake Alaotra area has increased from 109,000 in 1960 (Pidgeon, 1996) to 550,000 in 2003 (Andrianandrasana et al., 2005). The number of fishers operating in Lake Alaotra increased from approximately 1,000 in the 1960s to over 4,000 by 1989 (Wilmé, 1994) and continues to increase, adding further pressure to already heavily exploited and overexploited fish stocks (Vanden Bossche and Bernacsek, 1991; Pidgeon, 1996; Benstead et al., 2003). There may be 8,000 fishers currently operating in Lake Alaotra (H. Andrianandrasana, DWCT, pers. comm.). Annual fish catches declined from a peak of 4,000 tonnes in 1963 to just over 2,000 tonnes in the mid-1970s (Moreau, 1979b; Pidgeon, 1996). Production then rose again to almost 3,000 tonnes in 1990 (Pidgeon, 1996) and then decreased to approximately 1,900 tonnes by 2006; current production is approximately 1,100 tonnes per annum (J. Rabemazava, Service Régional de la Pêche et des Ressources Halieutiques, pers. comm.).

3.1.4 Invasive aquatic plants

Water lilies (*Nymphaea* spp.) that once covered large areas of the lake and marsh (Pidgeon, 1996) have largely disappeared, and more than 70% of the channels and open areas in the marsh are clogged by two invasive aquatic plants; water hyacinth (*Eichhornia crassipes*) and giant salvinia (*Salvinia molesta*) (Andrianandrasana et al., 2005). Both of these exotic species are known to degrade water quality (by reducing dissolved oxygen levels in the water, potentially causing fish mortality), clog waterways, and out-compete native plants (Oliver, 1993; Navarro and Phiri, 2000; Julien et al., 2001).

3.1.5 Environmental changes and other threats

The Alaotra wetland also faces threats from marsh burning, conversion of marsh habitat to rice fields, deforestation leading to soil erosion and siltation, climate change, eutrophication, and pollution, particularly run-off from adjacent irrigated rice fields where both pesticides and herbicides are used regularly during the growing season (Pidgeon, 1996; Copsey et al., 2009a). Fluctuations in water level also influence the fishery, and extent of drainage could be critical for tilapias because they typically breed or have nurseries in marsh habitat. The effects of deforestation on siltation, temperature, and photoperiod can adversely impact tilapia reproduction (Moreau, 1982). Temporary paddy-type rice fields are often prevalent in marsh habitat when water levels permit. These fields are constructed by barricading an area with water hyacinth to maintain sufficient water level via natural flooding; however, this also modifies the flow of water to other areas and negatively impacts habitat-fishery linkages (see Barbier, 2000).

3.1.6 Governance

The system of governance currently in place at Lake Alaotra is illustrated in Figure 3.2. External input and pressures from international agencies are not included in Figure 3.2; however, these sources are expected to influence decisions at national level (such as gazetting a new protected area). Restrictions to control fishing practices have been in place since the 1990s and an annual two-month fishery closure began in 2001, recently from 15 November to 15 January (Andrianandrasana et al., 2005). The

National Government implements the closure through regional authorities; the authority for Lake Alaotra is the Service Régional de la Pêche et des Ressources Halieutiques (Fisheries Service) based in Ambatondrazaka. Fishers in local communities are represented by a Federation of Fishers that works with the Fisheries Service to co-manage the fishery by developing, implementing, and monitoring fishing regulations. In practice, officers of the Federation of Fishers rarely visit fishing villages or interact with fishers; they also share an office with the Fisheries Service in Ambatondrazaka and are therefore viewed by many fishers as an extension of the Fisheries Service rather than a voice for local fishers. Consequently, despite the intention, the Federation is mistrusted by many fishers. The Federation also collects funds via license fees and a levy on fish sold to fish collectors. Twenty percent of these funds are meant to go to regional administration, 20% to the commune (i.e., group of villages), and 60% ultimately back to member fishers to assist them during the annual closed period for fishing. However, the legal structure establishing the Federation does not allow re-distribution of funds directly to individuals but only to institutions (DWCT Madagascar, pers. comm.). Accordingly, although the funds have been collected since 2005 they have not been used to benefit fishers, who perceive corruption to be the underlying reason and a major problem.

Although the Fisheries Service is responsible for patrolling the lake to monitor use of prohibited gear, and also illegal fishing activity during the closed period, this has largely ceased. The primary manner in which regulations are enforced is by monitoring trucks used by commercial fish companies, which are (a) not permitted to operate during the closed period and (b) randomly inspected at specific checkpoints in relation to limits for minimum fish sizes over the remainder of the year (B.J. Rasolonjatovo, DWCT Madagascar, pers. comm.).

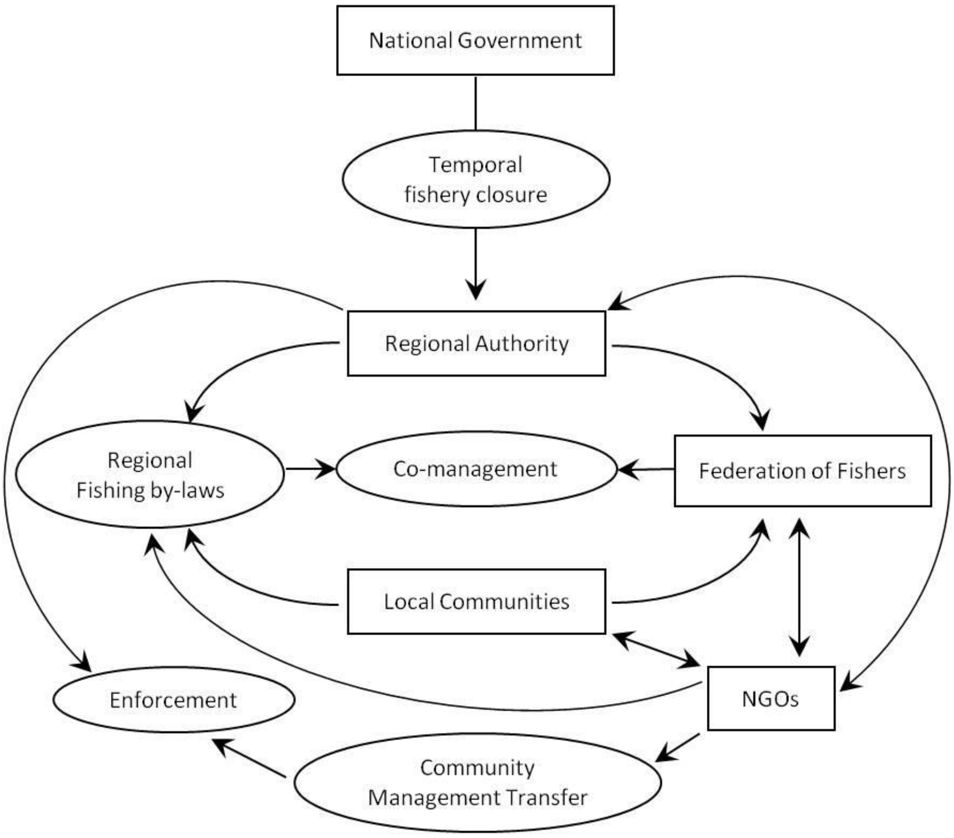


Figure 3.2. Diagram illustrating the governance system for the Lake Alaotra fishery. Rectangles represent organisations. Ovals represent measures. The diagram expands the governance system component shown in the conceptual framework diagram (Figure 2.3) in Chapter 2.

Regulations for the Lake Alaotra fishery are embodied in a code of practice known locally as the Dina de Pêche (Regional Fisheries By-laws) and include gear restrictions, minimum fish-size restrictions, wildlife laws, and fines for infringement (DWCT and Service de Pêche, 2002, 2006). These regulations are directed towards protecting marsh habitat for wildlife (particularly the critically endangered Alaotran gentle lemur, which is the primary interest and conservation goal of DWCT) as well as maintaining fish stocks and fish production for local livelihoods. The regulations generally reflect management objectives. For example, the blotched snakehead is an aggressive invasive fish that preys on endemic species as well as species of greater commercial value; therefore there are no restrictions on catching blotched snakehead because the management goal is to heavily exploit the species and thereby reduce its local population and impact on other species. Although tilapias, carp, and goldfish are introduced species and not conservation priorities, they are important from a livelihood perspective and regulations (such as minimum length and mesh size restrictions) are intended to protect them from overexploitation. The Madagascar rainbowfish is endemic to the lake but resistant to heavy exploitation due to physical refuge during the wet season when they are difficult to find and catch; although this species is a conservation concern, there are no practical measures to protect it. While the goals of stakeholders may appear to often conflict, DWCT also considers fishery sustainability to be of key importance. DWCT acknowledge that fishers and other local people must be partners in conservation and their livelihoods must be taken into account in conservation management; similarly DWCT expects that maintaining viable fisher livelihoods will reduce pressure to convert marsh habitat to rice fields and promote conservation and biodiversity goals.

Representatives of local community fishing groups have direct input into reviews of fishing regulations. Non-governmental organisations (NGOs) such as DWCT also have direct input into developing and amending regulations, and are involved in enforcing regulations through the Fisheries Service. An initial code of practice was agreed between stakeholders in 1998. This was formalised and extended as Regional Fisheries By-laws during a two-day meeting in Ambatondrazaka in July 2002 (DWCT and Service de Pêche, 2002), and then reviewed and amended during another two-day meeting in July 2006 (DWCT and Service de Pêche, 2006).

Since 2001 a series of workshops at regional, commune (i.e., group of villages), and village levels has further stimulated community involvement in conservation and management activities (Andrianandrasana et al., 2005). DWCT is working with all stakeholders to transfer greater management authority for the lake and marsh to local communities. The areas subject to management transfers are based on traditional community boundaries. Two legal instruments (Gestion Contractualisée des Forêts (GCF) and GEstion LOcale SÉcurisée (GELOSE)), have been used to arrange legal contracts between the commune, local resource users, and the relevant state agency regarding use and management of local lands. Currently, 35% of the marsh is under community management through contracts implemented by GCF and 10% of the marsh through GELOSE. It should be noted that not all components of the governance system function efficiently or effectively in all cases. For example, my research indicates that fishers and local communities perceive many inequities in the system and are unable to exercise their intended role. Many stakeholders also commented that the system is compromised by corruption. These issues will be explored in more depth in subsequent chapters.

3.1.7 Local conservation

Durrell Wildlife Conservation Trust (DWCT) has been conducting research on the endemic species of the Alaotra region since 1986, and since 1997 has promoted conservation to local communities through education and public awareness programs (Andrianandrasana et al., 2005). The critically endangered Alaotran gentle lemur (*Hapalemur alaotrensis*) is endemic to the Lake Alaotra wetland and confined to marshes adjoining the lake. Accordingly, the lemur and marsh habitat it depends on are of high conservation interest globally as well as locally, and have been the focus of considerable research and conservation effort (see Mutschler et al., 1994; Mutschler and Feistner, 1995; Mutschler et al., 1998; Mutschler et al., 2001; Mutschler, 2002; Waeber and Hemelrijk, 2003; Andrianandrasana et al., 2005; Ralainasolo et al., 2006; Guillera-Arroita et al., 2010b; Lahoz-Monfort et al., 2010; Hudson, 2011). In 2006 DWCT and regional authorities planned a series of zones to manage use of the Lake Alaotra conservation area (Figure 3.3). This was an initial step in developing a comprehensive management plan required by the Malagasy Government to designate the Alaotra wetland as a permanent protected area (Andrianandrasana, 2009). The plan consists of different management zones within the marsh, including a strict conservation zone for the Alaotran gentle lemur (where all forms of resource harvesting are prohibited), fishing-only areas, and regulated-use areas where fishing and some harvesting of marsh products are permitted (Figure 3.3).

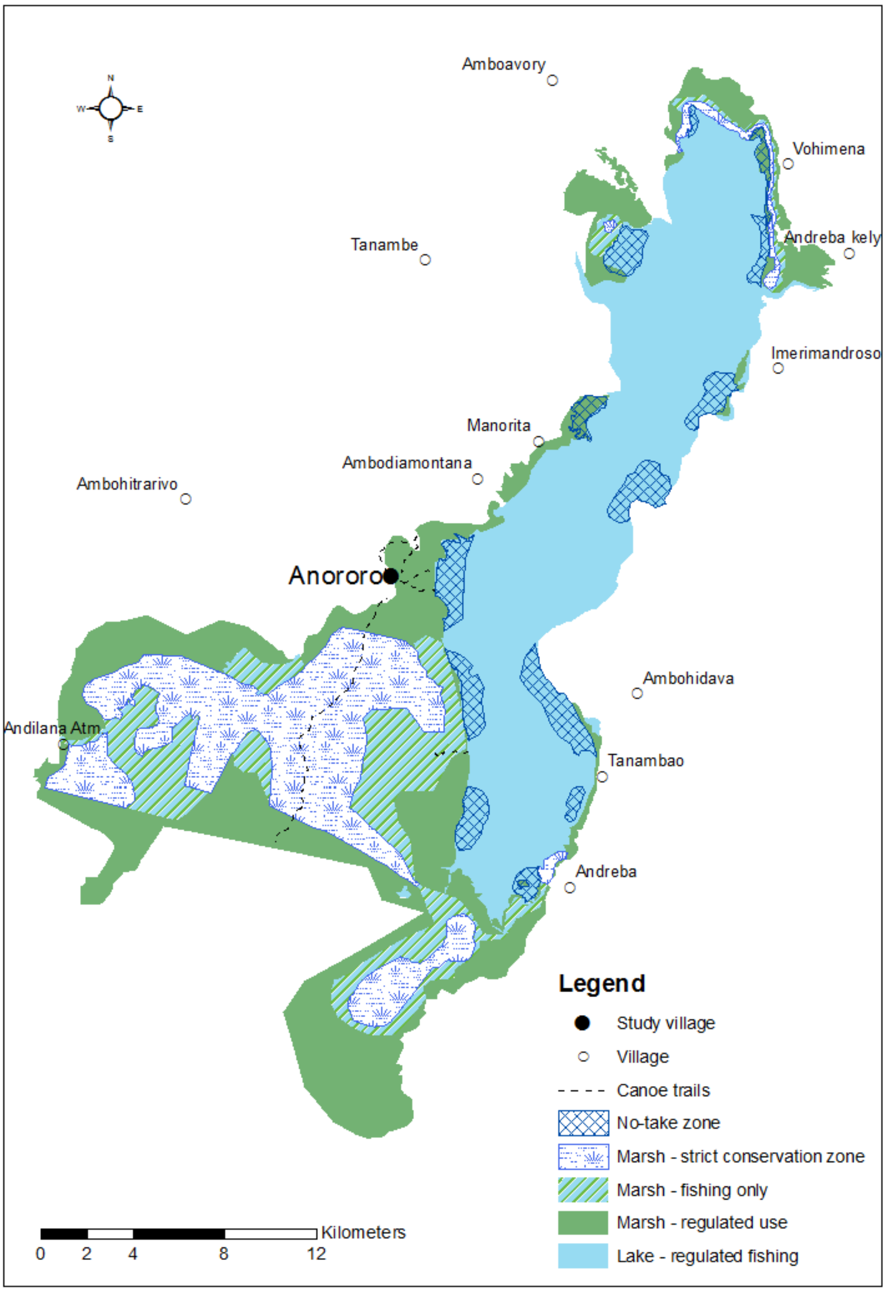


Figure 3.3. Map of Lake Alaotra showing planned management zones within the lake and adjacent marsh.

In addition to regulations relating to gear use, mesh sizes, minimum fish lengths, and a temporal closure, 12 no-take zones (NTZs) have been proposed for the lake to protect fish spawning grounds (Figure 3.3). The location of each NTZ was derived during a DWCT-sponsored study in December 2004 (Razanadrakoto and Rafaliarison, 2005). Even though the NTZs cover 15% of the surface area of the lake, to date there has not been any systematic research regarding the potential costs or benefits of these zones for local communities. While many local fishers are aware of some of the NTZs, implementation has been sporadic and little or no enforcement has occurred. Markers (primarily buoys and lines) were used to delineate the boundaries of NTZs; however, these were subsequently removed and presumed stolen, and those responsible for removal have not been formally identified (DWCT Madagascar staff, pers. comm.). Boundaries for the NTZs were then demarcated using reeds (*Phragmites communis*); five zones were demarcated in 2009, and it was planned to demarcate a further five zones in late 2010 and the remaining two in 2011. Three of the five NTZs demarcated in 2009 were dismantled by fishers shortly thereafter (B.J. Rasolonjatovo, DWCT Madagascar, pers. comm.). Based on implementation outcomes, low levels of compliance, and fisher feedback it is evident that an improved understanding of fishers' spatial behaviour and socioeconomic impacts for local people is required if the no-take zones and restricted-use areas are to be effective management options.

3.1.8 Study village

The base for the research was Anororo village (Figure 3.4), a community of approximately 8,000 people on the western edge of Lake Alaotra and administratively within the Alaotra-Mangoro region of Madagascar (PCD, 2004). The Anororo Commune encompasses five villages (*fokontany*) and covers an area of 176km^2; Anororo village is the largest of the five villages in the commune. The subsistence and cash economy of Anororo centres on fishing and rice cultivation as the primary livelihood activities. Anororo was selected for the case study because of characteristics facilitating the research and maximising the range of variables available for analysis: (i) close proximity to marsh and lake habitat, (ii) a large population of fishers using a wide variety of fishing habitats (to ensure a large sample representative of local habitat use), (iii) proximity to planned conservation interventions that could directly impact fishers, (iv) dependence on fishery and marsh resources for subsistence and commercial activity (to allow assessment of the socioeconomic impacts of a range of management interventions), and (v) use of a broad range of fishing methods (to examine spatial and temporal differences in fisher behaviour according to gear preferences). The main market for fish caught by

Anororo-based fishers is to commercial buyers travelling from Antananarivo, Madagascar’s capital; local market and bicycle collectors also operate in the village.

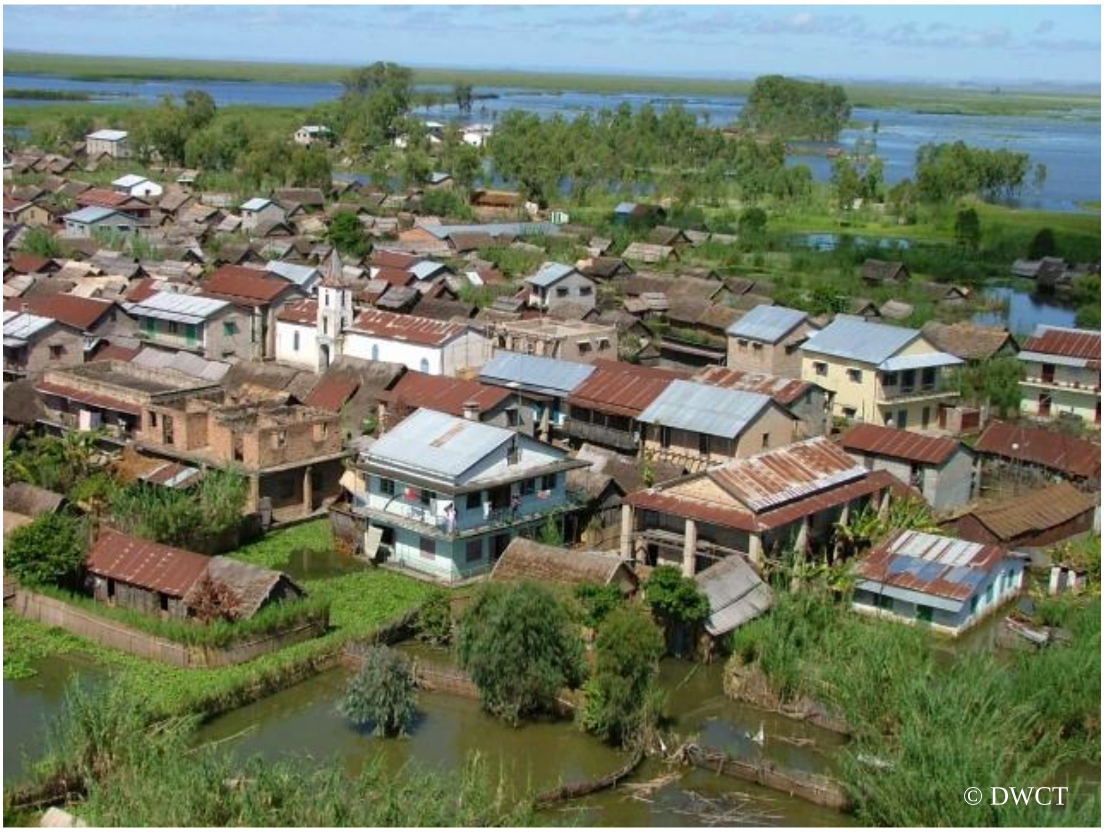


Figure 3.4. Aerial view of Anororo village during a major flood in 2005.

Although the majority of Anororo residents are from the Sihanaka ethnic group there are five other ethnicities represented in the village; in decreasing prevalence these are Merina, Betsileo, Betsimisaraka, Bezanozano, and Antandroy (Ralainasolo, 2004). Fishing is one of two primary livelihoods in Anororo; the other is rice cultivation. Secondary livelihoods include textile production and weaving using marsh products (primarily undertaken by women), teaching, operating a shop, and sewing, as well as transport occupations such as bicycle taxi services or running a small tractor and trailer (*kibota*). Anororo residents follow a traditional calendar based on a repeating 12-day cycle of work days (*andro tsara*) and days off (*andro ratsy*). The sequence is 5 work days, 1 day off, 2 work days, 1 day off, 2 work days, and 1 day off. The days off are intended to give rice farmers time for other activities such as fishing, and working the land (or with soil generally) is not permitted on these days (Jarosz, 1994).

Similarly, it was forbidden by taboo to use gill nets for fishing; however, when tilapia species became prominent in catches it became acceptable to use nets (Pidgeon, 1996).

Fishing in Anororo is generally considered a man's profession. Anororo women working in fishing tend to be fish collectors, either for commercial fish companies or local markets. There are four types of fish collectors operating in Anororo village: (1) bicycle, (2) local market, (3) Sandenda vola, and (4) Vazaha kely. The two commercial fish companies operating in Anororo, Sandenda vola since 1982 and Vazaha kely since 2003, engage collectors to buy fish from fishers on the lake and in the marsh; approximately 70% of these collectors are women. Only fish meeting minimum size limits are in turn purchased from collectors by the commercial companies (Bary-Jean Rasolonjatovo, DWCT, pers. comm.). Fish are not weighed or counted in any transaction but instead are measured in mixed-species bucketfuls using a 15L bucket. Fish are then transported on ice by truck to Antananarivo to be sold. Fish that are not sold to commercial collectors can be sold in a local market that operates daily in Anororo, or to collectors transporting fish by bicycle; these 'bicycle collectors' purchase fish from fishers on the lake, return to Anororo, and then ride several kilometres to sell fish in the markets of neighbouring villages.

This study builds upon previous research conducted in the area (see Moreau, 1979b; Andrianandrasana et al., 2005; Copsey et al., 2009a; Ferry et al., 2009; Guillera-Arroita et al., 2010b; Lahoz-Monfort et al., 2010) and the site characteristics are reasonably well known. DWCT also continue to have an active presence in the area through ecological monitoring and conservation education programs.

3.2 RESEARCH METHODS

This section is an overview of research methods and data collection techniques used during the study. Detailed methods and data analysis are described in each chapter.

3.2.1 Methodological approach

Although my academic background has primarily been in the natural sciences, I have worked on numerous conservation projects across Africa and North America, and have come to realise that conservation cannot succeed without involving people and accounting for their perspectives. This is particularly the case in developing countries where local people often rely on natural resources for livelihood. While I will always be an outsider in Anororo, I spent more than one year living in the village, speaking with and observing local people to gain insights and understanding of village life. I believe that taking the time to do this made it possible for me to be well-accepted within the village. Many fishers viewed me as a potential voice for them and stated that they believed authorities would listen to me. Shortly after I arrived in Anororo one of my field assistants questioned why '*vazahas*' (Europeans) cared so much about the lemur but not about them. This was an interesting question, similar to one I had also heard in Uganda, and it made me reflect on the purpose and potential value of my study at broader scale. As a PhD student partly funded by DWCT I had an obligation to provide results they could use to improve management and conservation of the Lake Alaotra wetland. At the same time I needed to have an open and objective outlook in order to identify weaknesses in DWCT's approach thus far and suggest possible solutions for the benefit of all stakeholders and their interests.

There is growing recognition within fisheries management that it is necessary to make management processes more participatory and transparent (Salas et al., 2007). This requires an interdisciplinary approach to research, particularly incorporating the social sciences (Berkes, 2012). There is an extensive range of quantitative and qualitative social research methods, which should be selected according to the specific research questions. Studies also often combine methods in order to triangulate information collected (Abernethy et al., 2007).

Following an introductory meeting and mapping exercise with fishers, three social research methods (focus groups, interviews, and participant observation) were most appropriate for my research; the advantages and disadvantages of each are described below. Fish length and weight measurements, water depth and water quality measurements, and opportunistic direct observations were also used for data collection. In all cases respondent codes were assigned to fishers to preserve their anonymity (Bryman, 2001; Bernard, 2002). Data were collected continuously between June 2009 and February 2011, except for August and September 2009. My field trips occurred in three phases. Phase 1 was a pilot investigation from June to July 2009, Phase 2 was from October 2009 to March 2010, and Phase 3 was from September 2010 to February 2011. Three primary research assistants continued to collect catch characteristic data and conduct fisher follows during my absence between Phase 2 and Phase 3.

3.2.2 Ethics

All data collection occurred with the informed verbal consent of participants. In order to introduce the project and request participation from the local community, an introductory meeting with 16 well-respected fishers identified by key informants was held at the beginning of the study on 18 June 2009. The purpose of the meeting was to communicate the aims of the project, voluntary role of participants, and expected project outcomes. All participants were encouraged to discuss the research with me if they had any questions or concerns.

Subsequent interactions with fishers (e.g., during focus groups, interviews, or fisher follows) included clear communication about the research. Fishers were again advised that participation was voluntary, they could withdraw at any time, and they were not required to answer any question that they did not wish to. Fishers who participated in the study also received a 'Participant Information Sheet' as a record of project aims and methods; this was reviewed and approved by the Imperial College Research Ethics Committee (ICREC) prior to commencing data collection. The information sheet emphasised that this was a student research project, confirmed that data records would be coded to ensure anonymity, and all information provided by fishers would be treated as confidential at all times.

Gaining the trust and confidence of fishers was integral to data collection. I employed local people as research assistants and lived with a host family in the village, which led to increased community support for the project and contributed to my credibility. On-water research activities were suspended for the duration of the fishery closure to further engender trust between the research team and fishing community; the team did not want to be perceived to be monitoring fishing activity during the closed period.

Throughout the study we reassured fishers that the information they provided as individuals would not be linked directly to them (to preserve individual anonymity) and would remain confidential within the research team. Although this was the case, despite the assurances some fishers within the community were still wary of our activities and did not want to participate in the research. As the study progressed and fishers observed that the research was being conducted in the manner I had described (i.e., studying fishing activity and catches, voluntary participation by fishers, not forcing fishers to answer questions, and not policing their activities), an increasing number of fishers approached the research team and asked if they could participate. This was taken as a sign of growing trust and confidence in the research. I was fortunate to have sufficient time to engender trust within the study village, which was invaluable for achieving many of the results of the study.

Although participation in the project was voluntary, fishers were provided with gifts in recognition of their time and contribution. Small gifts (such as photographs, or a *mofogasy* – a Malagasy cake) given to fishers for participating in interviews were of little monetary value (from £0.04 to £0.50). These were a gesture of goodwill and are believed to have not altered fishers' behaviour or responses. Fishers who participated in fisher follows were given larger gifts (such as raingear, a hat, or a headlamp) at the end of the project in recognition of their involvement throughout all aspects of data collection. These gifts were not anticipated by the fishers and hence are believed not to have influenced their behaviour or responses during the study. Although incentivizing participation in research may influence respondent behaviour and set unrealistic precedents for future researchers (Bernard, 2002), the small gifts I provided were deemed locally appropriate by field assistants, fitted well with local traditions and customs, and are not prohibitively expensive for researchers. I also consider it important that researchers acknowledge the value of the time and effort given by respondents to participate in research, particularly in developing countries.

3.2.3 Research team

The core research team consisted of three local assistants, myself, and my husband, Graham Wallace (Figure 3.5). Local research assistants and G. Wallace assisted with all aspects of project fieldwork and data collection under my direction. In addition, two field assistants were recruited to help with some aspects of data collection on an *ad hoc* basis, also under my direction. Malagasy is the common language used in the study area. French is the second most common language and English is not widely spoken and is rarely written. I am fluent in French and over the course of the study also acquired an intermediate knowledge of Malagasy. G. Wallace has intermediate knowledge of French and also developed some skills in Malagasy over the course of the study. All local assistants were fluent in French and Malagasy and were well-known in the village. Also, one fisher (and key informant) was recruited as our full-time paddler; additional paddlers were recruited as needed.

I developed the research questions, data collection protocols, and questions for semi-structured interviews and focus group sessions. My supervisors and colleagues provided feedback in accordance with normal academic practice. Local assistants provided guidance regarding cultural norms and local customs. Prior to data collection I trained each member of the team in interview and focus group techniques, the use of GPS units, and the data collection protocols required for fisher follows and recording catch characteristics. The roles of team members included assisting with organising village meetings and focus group sessions, conducting in-depth semi-structured interviews as well as short structured interviews with fishers, conducting fisher follows, and translating between French and Malagasy. Research assistants worked in pairs (usually with myself or G. Wallace) to facilitate efficient data collection and a team atmosphere. I led team meetings regarding project logistics and scheduling, which occurred weekly; all datasheets were submitted to me daily.

Figure 3.5. Members of the research team, other than myself. From left to right: Luhanaud Andriamiarivola, Narcisse, Joachin Randriarilala, Graham Wallace, Solofoniaina Esperant Rakotonisainana, Rado Zilia Randriamihamina, and Rabemanisa Emile. Ravo Andriamizana not pictured here.

3.2.4 Methods for data collection

3.2.4.1 Mapping exercise

We conducted a mapping exercise with a group of senior and experienced fishers to develop a preliminary map of the fishing area and key fishing locations. We used a blackboard and chalk to draw a map of the area and place fishing locations as fishers called them out (Figure 3.6). We later visited the primary landmarks and readily-identified locations nominated by fishers during the meeting to record and plot GPS tracks and waypoints (see Jones et al., 2008). This provided a locally relevant and geo-referenced base map of the fishing area. The map continued to be developed throughout the study as additional fishing locations were reported by fishers and as the locations became accessible for mapping. In order to determine the extent of each fishing location, mapping involved recording GPS waypoints at the northern, southern, eastern, and western boundaries as well as the central point of the location. In some cases, where only a central point was identified, local fishers indicated the extent of the boundary (e.g., 100m in all directions from the centre). A total of 92 fishing locations used by Anororo-based fishers were identified, and 81 of these were mapped.

Figure 3.6. Hand-drawn map of the fishing area and fishing locations nominated by fishers during the mapping exercise.

3.2.4.2 Focus groups

Focus groups are interactive group settings and an appropriate method for exploring issues at group level where participants have common interests and/or experiences forming a broad knowledge base that would be difficult to access through individual interviews (Krueger and Casey, 2000; Bernard, 2002). Focus groups are most effective when conducted to address relatively specific issues, and can generate a large quantity of data over a short period of time (Bryman, 2001; Esterberg, 2002). They are often relatively unstructured, and rely on group dynamics as well as the skills of the moderator to draw information from participants and summarise conclusions (Krueger and Casey, 2000; Esterberg, 2002). The function of the moderator is to invite input, ensure the discussion stays ‘on track’, and suggest further questions or different perspectives to be considered (Krueger and Casey, 2000; Bryman, 2001). Focus group sessions may also include participatory mapping or resource-distribution exercises to develop locally relevant maps and checklists (Smith, 2003; Chambers, 2006; Jones et al., 2008). One disadvantage of focus groups is the possibility of ‘groupthink’, where participants may ignore their own viewpoint and instead follow the group’s opinion (Janis, 1982); however, the debates and broad range of views expressed by fishers during focus group sessions suggests this was not the case in this study.

Focus group sessions were conducted to obtain background information about the fishery, locally-relevant seasons, patterns of fishing, equipment, catch characteristics, criteria for differentiating wealth, attitudes to local conservation, perceived fishery problems, possible solutions, and related issues. These sessions provided important baseline data, facilitated recruitment of fishers to the study, and built rapport with local people and community groups. We conducted five focus group sessions over the study period. Fishers from each of the eight village neighbourhoods, and using a range of fishing methods, were invited to attend; a total of 62 individual fishers (80% of invitees) participated.

3.2.4.3 Interviews

Structured (or standardised) interviews and questionnaire surveys are used to assess the attitudes and perceptions of local people about resource and conservation issues in a wide range of research contexts (see Infield, 1988; Newmark et al., 1993; Newmark et al., 1994; Ezealor and Giles Jr., 1997; Gillingham and Lee, 1999; Byers et al., 2001; Infield and Namara, 2001). Although these techniques are more likely than less-structured methods to yield data suitable for quantitative analysis (Bryman, 2001;

Bernard, 2002), they are relatively formal and inflexible, and restrict the interviewer's scope to explore responses further when interesting issues are raised (Bryman, 2001; Esterberg, 2002). This may limit the amount of information that respondents provide during interviews, especially for sensitive or contentious subject matter (Briggs, 1986). Semi-structured interviews involve a set of standardised questions as well as open-ended questions that promote further discussion (Bryman, 2001; Esterberg, 2002); they are particularly effective in multi-method research settings where relevant variables are likely to be interlinked (Ormsby and Kaplin, 2005). A further benefit of semi-structured interviews is that they are usually more participatory and interactive than other interview techniques (Bryman, 2001; Esterberg, 2002), which assists in building rapport with respondents and establishing the relevance of the research to their interests and concerns (Briggs, 1986; Arksey and Knight, 1999; Bryman, 2001). The approach also accommodates and accesses the substantial body of knowledge held by local resource users. Both structured and semi-structured interviews were used during the study.

3.2.4.4 Catch-monitoring interviews

Participatory catch assessment techniques may be used to evaluate the present state of a fishery and involve direct sampling of fisher effort and catches made (Welcomme, 2001). Structured catch-monitoring interviews (herein 'catch interviews'), including direct measurement of fish lengths, were conducted opportunistically with Anororo-based fishers who were returning home at the end of their fishing trip. Fishers were interviewed regardless of type of gear used, size of catch, species caught, or fishing location used, in order to maximise sample sizes as well as spatial variation within the dataset, and to increase overall understanding of the fishery.

Anororo-based fishers use (i.e., leave from and return to) 12 landing sites around the edge of the village. Catch interviews were conducted on the lake and in the marsh prior to fishers selling their catch to collectors (i.e., fishers do not return to the village with their catch if they intend to sell it, which was the case for most fishers). Catch interviews were conducted at two places each week; site 1 was 2km east of the village at the lake-marsh edge and site 2 was in an open area of the marsh approximately 500m south-west of the village. These sites were situated in the main thoroughfares used by fishers when returning to the village from all fishing locations and were also where collectors waited to buy fishers' catches. Fishers from all 12 landing sites in the village were interviewed.

Catch interviews involved asking fishers a series of questions about their fishing activity for the day, followed by counting and measuring the fish they had caught (see Appendix S3.1). Fishers were asked to provide: (i) the name of the landing site they used at the village, (ii) the local name of the fishing location where they had made their catch, (iii) the time and distance to travel to and from their landing site and the fishing location, (iv) time spent fishing, and (v) details of the gear they used, such as quantity, size, mesh size, and the set and retrieval time. Fishers were also asked whether they were going to sell all or some of their catch and/or keep all or a portion for food, and to specify which fish would be sold or kept. Prices received for full or partial catches were obtained from fishers and/or collectors whenever possible (n = 258). Catch interviews were short, requiring between 5 and 10 minutes to conduct, and collectively provided a comprehensive understanding of fisher behaviour and returns from fishing within a relatively short period of time. The interviews were also designed to be readily useable by fishers within a participatory monitoring context in the future, facilitating greater levels of stakeholder involvement and larger sample sizes for monitoring data.

We conducted 1,800 catch interviews over the study period. A total of 537 individual fishers were interviewed, which is approximately 70% of the estimated total number of fishers based in Anororo; 289 fishers were interviewed once, 88 were interviewed twice, 44 were interviewed three times, and 116 were interviewed more than three times. Fifty-three percent of interviews were conducted by me or G. Wallace with assistance from a local research assistant; the remaining 47% of interviews were conducted by two local research assistants working together. Most fishing activity occurred very early each morning so that fishers could return and sell their catch to a collector who then required sufficient time to transport the fish for sale elsewhere. Consequently, the majority of fishing activity was finished by 1100h and few fishers were observed returning from fishing in the afternoon. For this reason, catch interviews were mainly conducted in the morning. Afternoon interviews were conducted once every two weeks; these were only conducted at site 1 because fishers were not observed to return from fishing during the afternoon at site 2.

Area-catch methods, such as catch per unit effort (CPUE), can be applied to catch data to estimate relative stock abundance (Sutherland, 2000; Welcomme, 2001). CPUE comprises two key variables of fishing activity (catch and effort) and can be defined as the quantity of catch for a specified amount of effort (Welcomme, 2001; SEAFDEC, 2005). Catch and effort information can provide an index of total catch and catch characteristics for a particular fishing gear or for a fishery (Welcomme, 2001; SEAFDEC, 2005). Catch is relatively easy to assess and comprises the quantity of fish caught by number or by weight. Calculation of effort is more complicated, particularly for artisanal fisheries where a diverse range of gears and fishing methods are used (Welcomme, 2001). Effort can be measured in terms of the quantity and dimensions of gear used, number of fishers or boats, and/or time (usually in hours) spent fishing (SEAFDEC, 2005). Estimates of catch and effort for each type of gear used are required to estimate the total catch of the fishery (Welcomme, 2001). CPUE is generally significantly lower in open-access areas than where access is restricted (Eggert and Lokina, 2010). However, the assumption that CPUE is directly proportional to abundance is known to be incorrect (Hilborn and Walters, 1992) and additional evaluation methods should be considered to improve estimate reliability. In this research, CPUE was primarily used to compare returns for fishers using the same type of gear in different locations (e.g., inside compared to outside no-take zones) as well as across seasons.

Anororo-based fishers can readily distinguish between the species of fish they catch. Although we used vernacular names for data collection, each species was photo identified by freshwater fisheries expert P.V. Loiselle. In some cases it was difficult to confirm species identity because two species known to be present in Lake Alaotra – Mozambique tilapia (*Oreochromis mossambicus*) and Nile tilapia (*Oreochromis niloticus niloticus*) (Moreau, 1979b) – have been reported to hybridize in other areas (de Silva and Ranasinghe, 1989; de Silva, 1997; Gregg et al., 1998). Such hybridization frequently occurs when allopatrically distributed species are put into sympatry in a new environment (P.V. Loiselle, WCS, pers. comm.). This phenomenon is also known to occur in Lake Itasy, Madagascar (Daget and Moreau, 1981). Hybrid tilapia in Lake Alaotra could be a mix of several species and in cases where species identity could not be confirmed a generic ‘tilapia’ category was used instead.

Species and length of individual fish caught were recorded during catch interviews, and length measurements were obtained for 27,064 fish. We did not collect data on fish lengths during 39 interviews because fishers had already sold their catch, did not want to have their fish measured, did not have time for their fish to be measured, or, for six interviews, had used dip nets to catch the tiny Madagascar rainbowfish (*Rheocles alaotrensis*). When catches were large (i.e., in excess of approximately 50 fish) a random sub-sample of the total catch was measured to minimise time demands on fishers (see Davies et al., 2009). This involved selecting an appropriate scale of sampling frequency, which depended on the size of the catch as well as the willingness of the fisher to wait while his fish were measured. The selected sampling frequency (for example, every second, third, tenth, or fifteenth fish) was recorded. Samples of these catches were believed to be representative of the entire catch because (a) fishers do not separate the species or size of fish within their catch before or at the time of interview and (b) research assistants were trained to select the next fish in line without looking, rather than ‘targeting’ a particular fish. The number of fish remaining after sampling at the selected frequency was also recorded to permit calculation of the total count of fish caught (i.e., number of fish measured × sampling frequency used + number of fish remaining = total catch). Sub-samples of catches were then used to extrapolate out to the total catch, to better represent catch compositions and length-frequency distributions as well as to calculate total catch weight. This produced a dataset of species and lengths for 63,936 fish.

Fish lengths were measured using a locally-constructed apparatus. A 60cm stainless-steel ruler was mounted onto a horizontal wooden base; at the 0cm end a piece of wood was attached at 90 degrees to the base (Figure 3.7). The mouth of the fish was placed against the vertical end of the board and the length of the fish was recorded to the nearest millimetre. All fish were measured in terms of total length (TL), except for common carp (*Cyprinus carpio carpio*) and goldfish (*Carassius auratus auratus*) where fork length (FL) was measured. Length measurements were subsequently converted to centimetres to estimate weight based on species-specific length-weight relationships.

Figure 3.7. Locally-constructed fish measuring board.

3.2.4.4.1 Length and weight measurements and relationships

Although it was not possible to record the weights of all fish measured during catch interviews (this would have been an excessive demand on fishers' time), a sample of fish obtained from fishers and fish collectors were measured and weighed separately to determine species-specific length-weight relationships. A sample of 498 fish lengths and weights were measured on 12 separate occasions between 4 November 2009 and 12 November 2010. Fish lengths were measured using the methods described above and converted to centimetres to facilitate calculation of length-weight relationships. A kitchen scale marked in 10g increments and with 3kg capacity was used to weigh fish to the nearest 5g. Very small fish of the same length were weighed together for a combined weight; individual weight was then estimated by dividing the combined weight by the number of fish weighed. More-sensitive handheld drop-weight scales were also tested; however, it was difficult to obtain accurate readings because the spring in the scale moved continuously due to canoe instability even when stationary on water. Although less sensitive, the sturdier kitchen scale provided more-consistent weight data and was easier and faster to use. The resultant length-weight equations were used to estimate the weight of each fish measured during catch interviews as well as the total weight of a fisher's catch.

3.2.4.5 Semi-structured interviews

3.2.4.5.1 Background interviews

Background interviews (see Appendix S3.2) were trialled during Phase 1 and revised for use during Phase 2 of the fieldwork. These interviews collected information about each participant's background, reliance on fishing for livelihood, fishing behaviour, income from fishing and other sources, perceptions of the status of the fishery, awareness of fishing regulations, and perceptions of the impacts and/or benefits of regulations. Interviews were conducted between 8 November 2009 and 30 January 2010, overlapping the fishery closure between 15 November 2009 and 15 January 2010. Each background interview comprised a set of core questions regarding information required from all respondents (Bryman, 2001; Esterberg, 2002) as well as open-ended questions that provided opportunities to explore interesting and/or unusual responses, including issues that respondents raised as being particularly important for them (Arksey and Knight, 1999; Bernard, 2002; Holmes, 2003).

Background interviews were conducted opportunistically by walking through all areas of the village and approaching people who had some involvement in fishing and were willing to participate. Fishers who had already participated in catch interviews and/or follows were also interviewed to permit triangulation of responses with observations as well as data for catches and fishing effort (Abernethy et al., 2007). Interviews were then arranged for times that were convenient for fishers. Fishers were interviewed separately (i.e., without other fishers present) at their home to maximise the power of interviews and integrity of quantitative data for analysing fishers' behaviour (see Jones et al., 2008). After an initial training period, interviews were either conducted by or translated by one of three research assistants. Reliability tests were conducted on two occasions by comparing translation of the same interviews by each of the three research assistants; one-way analysis of variance (ANOVA) confirmed that there were no significant differences between the assistants in interview translation (Test 1, $df = 2$, $F = 1.21$, $p = 0.30$; Test 2, $df = 2$, $F = 1.01$, $p = 0.36$). We conducted a total of 405 background interviews with people who had some involvement in fishing, irrespective of whether their involvement was full-time, part-time, or on a casual (i.e., itinerant) basis; all respondents were men. Each interview required approximately one hour to conduct.

3.2.4.5.2 Scenario interviews

Semi-structured interviews during Phase 3 of the fieldwork included scenario analyses (Cinner et al., 2008). Scenario interviews were conducted 11 to 12 months after the background interviews conducted during Phase 2, and only with fishers who had participated in background interviews in order to permit triangulation of responses. Scenario analysis is based on developing plausible and meaningful scenarios about possible future circumstances to explore the consequences of management decisions (Chen et al., 2003; Damania et al., 2005; Islam and Braden, 2006). The scenarios are then discussed with resource users to determine their perceptions of how the circumstances would affect their behaviour and livelihood (Hoang Fagerström et al., 2003). This information can then be incorporated into conservation planning to provide insights about how management strategies may impact local people (Chen et al., 2003; Milner-Gulland and Rowcliffe, 2007). Scenario analysis provides a useful framework for understanding the socioeconomic importance of artisanal fisheries (Cinner et al., 2011) and enables the costs to fishers of alternative conservation scenarios to be estimated prior to implementation. Because the method engages resource users and takes their interests and concerns into account it can ensure that management decisions are locally-appropriate and minimise costly mistakes or revisions (Iachetti, 2007).

Scenario interviews (see Appendix S3.3) were used to assess fishers' perceptions of the effects of various management interventions and changes in fishing intensity on fishing behaviour and returns. Scenarios addressed perceptions about the status of the fishery, how fishers would be affected and respond if some areas were closed to fishing, how their fishing effort or spatial behaviour would change if some areas were closed or restricted for fishing, where closed or restricted areas could be best located and why, and considerations to be taken into account when designing and implementing closed or restricted areas. The same information was collected in relation to the annual closed period for the fishery to permit comparison of impacts and responses between interventions.

Scenario interviews were conducted between 2 October and 16 December 2010. After an initial training period, interviews were either conducted by or translated by one of two local research assistants. We conducted 221 scenario interviews in total. Each interview required approximately one hour to conduct.

3.2.4.6 Fisher follows – participant observation

Participant observation conducted during fieldwork may provide quantitative as well as qualitative data (Bernard, 2002). The key prerequisite for effective participant observation is gaining sufficient trust from people to be able to observe their behaviour from close proximity without modifying their behaviour (Bryman, 2001; Esterberg, 2002). Trust lowers levels of reactivity by those observed and eventually allows monitoring of participants' usual activities to maximise the validity of data collected (Bernard, 2002). Participant observation is frequently used in conjunction with questionnaire, survey, or interview-based methods and is particularly useful to triangulate information derived from other sources (Esterberg, 2002; Abernethy et al., 2007). Drawbacks to this method are cost in terms of research time and relatively small sample sizes because it is usually not feasible to observe a large number of people in detail in conjunction with other data collection.

Fisher follows (see Appendix S3.4) were conducted to corroborate catch monitoring data as well as data from both sets of semi-structured interviews. They also provided opportunities to 'gain a feel' for daily fishing activities and constraints. Because extended or multi-night follows of fishers were not feasible or safe, follows focused on fishers who conducted daily trips from Anororo. Followed fishers travelled up to 22km (one way) on a daily fishing trip, and covered a wide range of habitats and fishing locations throughout the lake and marsh. Even though fishers who made extended fishing trips spanning several days were not followed, we were able to account for their fishing activity because they sometimes participated in catch interviews on their return to the village. Not following these fishers is therefore expected to have not compromised the range of data collected.

Forty Anororo-based fishers collectively using a range of fishing locations, habitats, and types of gear were identified by key informants and recruited to participate in fisher follows. To minimise demands on each fisher's time they were invited to be followed once in each two-month period over the full calendar year (i.e., six follows per fisher per year) to account for temporal and seasonal variation in fishing behaviour. Fishers were also requested to advise the research team of any notable changes to their fishing patterns and/or effort (such as switching gear and/or fishing location); additional follows were conducted if this occurred. Of the 40 fishers recruited, 27 (67.5%) used traps, 10 (25%) used gill nets, 2 (5%) used cast nets, and 1 (2.5%) used the line & hook method; these proportions were similar to those reported during catch interviews for the major sample of fishers. Five of the followed fishers used traps and gill nets, depending on time of year; four trap fishers also used gill nets and one gill net fisher also used traps. Fishers using the '*jinjira*' (rod & bubble hand method), '*mangodo*' (slap & bubble hand method), or '*sitra*' (dip net), which are prohibited under the 'Dina de Pêche' (Regional Fisheries By-laws) were willing to participate in catch interviews but did not want to be followed.

Because it was not feasible to travel in a fisher's canoe, we conducted follows using either rented or project-owned canoes. Details recorded during each follow were (i) landing site used, (ii) time of departure from the landing site, (iii) time of arrival at the fishing location (time fishing activity commenced), (iv) fishing method used, (v) time fishing activity ceased, (vi) quantity of gear items used, and (vii) time of arrival back at the landing site. A hand-held tally counter was used to count the quantity of gear items used or the number of times a cast net was cast. Garmin eTrex Legend HCx GPS units were used to record spatial information for the fishing location as well as locations of gear items. Waypoints were recorded at the first and last trap, line & hook, or throw of a cast net, as well as every tenth trap, line, or throw between the first and last. This procedure of recording a waypoint for every tenth gear item meant that research activities did not delay the fisher but still provided an accurate count of items and representation of spatial fishing behaviour. For gill nets we recorded a waypoint at each end of each net to permit calculation of net lengths.

A catch interview was conducted with the fisher at the end of each follow, before any fish were sold to a collector. Research assistants were trained to not disclose to the fisher any information gathered during the follow that could influence the fisher's responses to interview questions. Due to the timing of fishing activity (i.e., often in the middle of the night and/or very early morning), follows were pre-arranged with fishers one to several days in advance. Unfortunately, eight follow fishers left the project prior to completion and therefore six follows per fisher was not always possible. These fishers left the project because they moved from the village to camp near a better fishing location, stopped fishing due to illness, were pursuing other activities such as rice cultivation or making fishing equipment, or were waiting for fishing conditions to improve.

We conducted a total of 183 fisher follows over the study period. A median of 4.5 follows (range 1 to 10) were conducted per fisher (n = 40). Follows lasted for a median of 3.9h (range 0.58h to 10.5h) and totalled 768h of data collection. More than half (54%) of the follows were conducted by either myself or G. Wallace with one local research assistant. We do not believe that conducting follows changed or influenced fisher behaviour or fishing success. Fishers requested members of the research team to not carry any money on a follow because they believed the presence of money would result in them having a small catch (i.e., if you have money then you don't need a large catch). Some fishers even commented during a follow that we were 'good luck' because they were catching many fish.

3.2.4.7 Water depth and quality

Lake Alaotra is part of an extensive rainfall catchment area and water depth throughout the lake and marsh fluctuates over the year in line with increases and decreases in rainfall. To track variation in water depth throughout the year, we recorded weekly depth measurements at the two sites used for catch interviews. Measurements were always taken in the morning at the start of a catch interview session. Depth was measured using lead weights attached to one end of several metres of monofilament fishing line coiled around a buoyant section of bamboo. The line was uncoiled into the water until the weights struck bottom and no further tension could be felt in the line. The line was brought taut and then marked with a clothespin where it met the surface of the water. The length of the line from the weights to the clothespin was measured to the nearest centimetre and recorded as water depth. The lightweight fishing line used for the measuring device reduced drag when measuring water depth on windy days.

To gauge any differences in water quality that might influence catches across habitats, a range of parameters (temperature, dissolved oxygen, pH, and turbidity) were sampled twice at each of 15 fishing locations in the marsh (7), lake-marsh edge (6), and lake (2) between January and March 2010. Water samples were collected at an average depth of 45cm (approximately an arm's length) using a 500ml container with removable lid. The lid was replaced immediately after collecting the sample to prevent alteration of dissolved oxygen content. Using a CHEMets Dissolved Oxygen Test Kit and allowing time to ensure the canoe was sufficiently stable, oxygen content was measured for a range of dissolved oxygen values between 1ppm and 12ppm within five minutes of sample collection. Temperature was measured using a Hanna Instruments Checktemp 1 digital thermometer with stainless steel probe, ±0.3°C accuracy with calibration check. The pH of the water was measured using a Hydrion Lo-Ion pH Test Kit for a range of pH values between 5.0 and 9.0. Turbidity was measured using a black & white Secchi disk 20cm in diameter. The Secchi disk was lowered into the water until it just disappeared and the line was marked with a clothespin where it met the surface of the water. The Secchi disk was then raised until it just appeared and the line was marked with a second clothespin where it met the surface of the water. The two depths were measured to the nearest millimetre and the mean value was recorded as turbidity (i.e., Secchi disk transparency).

3.2.4.8 Direct observations

In addition to systematic data collection described above, further information about many aspects of the fishery, other livelihood activities, and life in general in Anororo village was obtained via opportunistic conversations and direct observations during the study. This included discussions with fish collectors at catch interview sites while waiting for fishers to return with their catch, as well as with research assistants and local rice farmers, including the host family. Fishers also offered unsolicited information during follows and catch interviews regarding problems with the fishery, why a certain fishing method or location was used, and/or recent events. This information provided valuable insights about how the fishery functions and the lives of people in Anororo, which helped frame research questions and formal data collection.

CHAPTER 4 CHARACTERISING THE LAKE ALAOTRA FISHERY: IMPLICATIONS FOR MANAGEMENT

4.1 INTRODUCTION

Inland fisheries are an important source of food and income for subsistence and artisanal communities in many areas of the world (Welcomme, 2001; Canonico et al., 2005). They accounted for 10.3% of the total world fish capture in 2009, and 90% of this catch occurred in developing countries (Welcomme, 2011b). Despite their importance, inland fisheries are understudied; however, the resources of most are believed to be fully utilised or overexploited (see Mölsä et al., 1999; Balirwa et al., 2003; Hilborn et al., 2003; Matsuishi et al., 2006; Lorenzen, 2008; FAO, 2010; Welcomme et al., 2010). The dispersed and informal nature of many inland fisheries leads to considerable underreporting of catches and fishing activity (Welcomme, 2011a). As a result, the role of inland fisheries in providing food security and protein for rural communities is often greatly underestimated (Pauly et al., 1998; FAO, 1999; Caddy and Garibaldi, 2000; Allan et al., 2005).

Most tropical inland fisheries are small-scale, multi-species, and multi-gear fisheries (SEAFDEC, 2005) that typically discard little or no fish (Jacquet and Pauly, 2008). Often described as artisanal or subsistence fisheries (Berkes, 2003), their primary characteristics are that fishers: (1) are poor, (2) harvest resources personally, (3) live close to the resource, (4) have a limited fishing range, (5) use low-technology gear, (6) fish as part of a long-standing cultural practice, and (7) collect resources that have a relatively low cash value (Hauck, 2000; Branch et al., 2002; Cardoso et al., 2005). Artisanal fishers can be distinguished from subsistence fishers according to their main use of the resource; whereas subsistence fishers use the resource only for food, artisanal fishers use their catches partly for subsistence and partly for income through commercial sale (Cardoso et al., 2005).

With appropriate management, artisanal fisheries can provide a sustainable source of protein and income for local communities. However, these systems are often data-poor, fish ages and catch profiles are rarely known, and management decisions are usually made with insufficient knowledge of catch sizes, fishing effort, and length-weight relationships (Welcomme, 2001; Holland and Herrera, 2009). To manage a fishery effectively it is critical to understand how the fishery functions, including who is involved, where fishers fish, how fishers fish, what fishers catch, and the effort involved in doing so. This information is particularly lacking for artisanal fisheries in developing countries. Further research is required to explore factors affecting inland artisanal fisheries and address current knowledge gaps (Ratner et al., 2004; Welcomme et al., 2010; Welcomme, 2011a).

The case study for this research is Lake Alaotra, Madagascar, where, like in other artisanal fisheries, fishing is fundamental to the subsistence and livelihoods of many people. Similar to other freshwater lakes in developing countries (see Balirwa et al., 2003), the composition and commercial focus of Lake Alaotra's fishery has varied in line with successive fish introductions. Prior to 1955, when redbreast tilapia *Tilapia rendalli* was introduced, common carp *Cyprinus carpio carpio* (introduced in 1926) was the dominant species; by 1958 *T. rendalli* comprised the majority of fish catches (Moreau, 1979b). There have been a series of fish introductions since the 1960s, each resulting in an eventual change of dominant species (see Kiener, 1962; Moreau, 1979b). Despite this history of introductions as well as occasional re-stocking programs, systematic efforts to manage the fishery for the ongoing livelihood of local people, long-term sustainability, and conservation of wetland habitat are relatively recent (Durbin et al., 2003; Ranaivonasy et al., 2005). Introduced and invasive species are a major threat to freshwater biodiversity in Madagascar (Benstead et al., 2003), and overexploitation of inland fisheries compromises the sustainability of current levels of reproduction and recruitment (Welcomme, 2011b). Annual fish production for the Lake Alaotra fishery has been declining for more than three decades (Figure 4.1; Moreau, 1979b; Pidgeon, 1996; J. Rabemazava, Service Régional de la Pêche et des Ressources Halieutiques, pers. comm.); the fishery continues to be under extensive pressure and requires effective management and conservation action if it is to remain viable (Durbin et al., 2003).

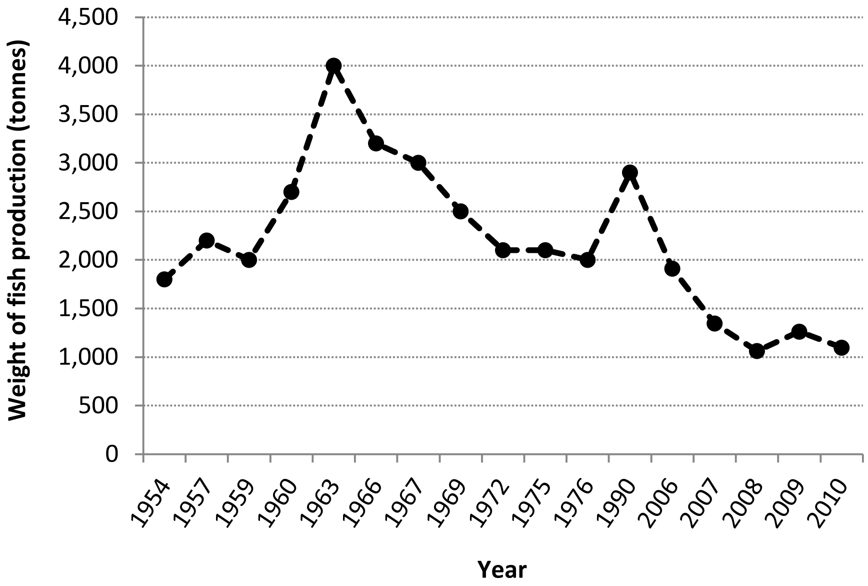


Figure 4.1. History of recorded fish production in Lake Alaotra, Madagascar. Data source: Moreau, 1979b; Pidgeon, 1996; J. Rabemazava, Service Régional de la Pêches et des Ressources Halieutiques, pers. comm.

Existing and planned management interventions aimed at achieving a sustainable fishery include different conservation zones within the marsh, regulated fishing and no-take zones in the lake (see Figure 3.3 in Chapter 3), an annual two-month fishery closure, stock enhancement, as well as regulations relating to gear use, mesh-size restrictions, and specified minimum fish lengths (see Ranaivonasy et al., 2005; Razanadrakoto and Rafaliarison, 2005; Dina de Pêche, 2006). However, many fishers have indicated that management interventions are inadequate to protect the fishery and do not correspond to prevailing conditions. Furthermore, the proposed management plan lacks information about local stakeholders' social and economic interests, and evidence of non-compliance has resulted in a plan that is largely 'on paper'.

It is critical to have a detailed understanding of the fishery and fisher behaviour in order to develop an effective management plan (Holland and Herrera, 2009). This study is the first characterisation of the Lake Alaotra fishery since Moreau's research in the mid-1970s (see Moreau, 1979a, b). I explored fishing distribution and effort, methods used, catch sizes and composition, as well as the lengths of individual fish caught to provide an understanding of the dynamics of the fishery over the calendar year. These data are common indicators of ecosystem health and the sustainability of yield from the fishery, and can be used to monitor changes in the status of the fishery over time (Welcomme, 1999; Powers and Monk, 2010). Subsequent chapters will expand on why fishers behave as they do and how this information could inform current and future management decisions.

4.2 METHODS

4.2.1 Study site

Please refer to Section 3.1 in Chapter 3 for details of the study site.

4.2.2 Data collection

Please refer to Section 3.2 in Chapter 3 for an overview of general data collection methods. Methods specific to this chapter are provided below.

Data were collected during June and July 2009 and for 14 consecutive months from October 2009 to December 2010, comprising two dry seasons (2009 and 2010) and one wet season (2010). I conducted structured catch interviews (n = 1,800), semi-structured background interviews (n = 405), semi-structured scenario interviews (n = 221), and fisher follows (n = 183) with Anororo-based fishers to collect detailed information regarding fisher demographics, catches (including catch size, species, and length of fish caught), habitats and fishing locations used, fishing methods used, and fishing effort. These data provided a comprehensive understanding of the status and dynamics of the Lake Alaotra fishery; a total of 784 fishers participated in one or more forms of data collection. Focus group sessions were used to corroborate data from individual interviews and provide additional contextual information. Respondent codes were assigned to all participating fishers to preserve their anonymity (Bryman, 2001; Bernard, 2002).

4.2.3 Data analysis

Statistical analyses using SPSS version 18.0 and R version 2.13.1 (R Development Core Team, 2011) explored relationships between fishing effort, gear use, habitat use, and catch characteristics (catch composition, fish sizes and weights, location where caught, and fishing method used) on both spatial and temporal scales, including seasonal and within season (monthly) variation. When catches exceeded approximately 50 fish a random sub-sample was counted and measured to minimise time demands on fishers (Chapter 3). These sub-samples were extrapolated to calculate total catch, which ensured a more accurate assessment of catch compositions, length-frequency distributions, and total catch weights, particularly when catches comprised hundreds of juvenile tilapia.

Fishing locations occurred within a range of habitats, which I categorised as marsh, lake-marsh edge, lake, and village border. Locations were considered to be in the marsh if they were 500m or more into the marsh from the edge of the lake. Marsh locations also included open bodies of water inside the marsh. Locations were considered to be at the lake-marsh edge if they were within 500m of either side of the edge. Lake locations were 500m or more into the lake from the lake-marsh edge. Use of 500m as the criterion to distinguish habitats was based directly on divisions described by Anororo fishers. Village border locations directly adjoined Anororo village (for example, adjacent to the main road). To account for the difference in the length of the wet season (5 months) compared to the dry season (7 months), the mean number of fishers that reported fishing within each habitat per month during each season was used for statistical analysis.

Length-weight relationships for fish species encountered in catches were fitted for each species using the equation: $W = a \cdot L^b$, where W is weight in grams, L is length in centimetres, and a and b are constants. Because of known hybridisation between species of tilapia (Daget and Moreau, 1981; de Silva and Ranasinghe, 1989; de Silva, 1997; Gregg et al., 1998), a generic length-weight relationship for all tilapia was also calculated. The derived length-weight equations were used to estimate fish weight from the length of each fish measured during catch interviews as well as the total weight of a fisher’s catch. The generic relationship for tilapia was used in cases where species identity could not be readily confirmed. Published length-weight relationships were used to calculate the weight of fish when sample sizes were insufficient to calculate a trendline.

I used a general linear model to determine the factors influencing the price fishers received for their catch. Explanatory variables included in the final model were gear type, habitat, season, weight of each species sold, total weight of fish sold, and the type of collector.

Analysis of fishers' perceptions of the state of the fishery was based on data from background (n = 405) and scenario (n = 221) interviews. Fishers were asked during background interviews how they perceived catch and fish sizes to have changed from 2005 to 2009 and the possible reasons for these changes. During scenario interviews, fishers were asked for their view on the current state of the fishery and reasons for this. Responses were categorised into common themes and presented as the percentage of fishers providing each response. Response sample size varies according to whether background or scenario interviews were used to collect the data and because fishers sometimes gave vague or ambivalent responses that could not be categorised explicitly.

4.3 RESULTS

4.3.1 Fisher demographics and fishing context

Fishers participating in semi-structured interviews ranged from <15 to 55+ years of age (Figure 4.2) and had a median of 11 years of fishing experience ranging from 1 to 60 years (Figure 4.3); 52% of respondents had in excess of 10 years' fishing experience. Years of fishing experience was strongly positively correlated with fisher age category ($r = 0.81$, $p < 0.0001$). The majority of respondents (96%, n = 405) were born and raised in Anororo, and almost all (99.5%) were members of the *Sihanaka* ethnic group; 0.5% were *Merina*. Of 405 fishers interviewed, 86% (n = 349) considered their primary occupation to be fishing. Whereas 31% (n = 125) of fishers relied entirely on fishing for livelihood, the majority (69%; n = 280) had an alternative source of income at some point during the year; stated sources of this income were working for another person in rice fields, operating a shop, growing vegetables, or sewing.

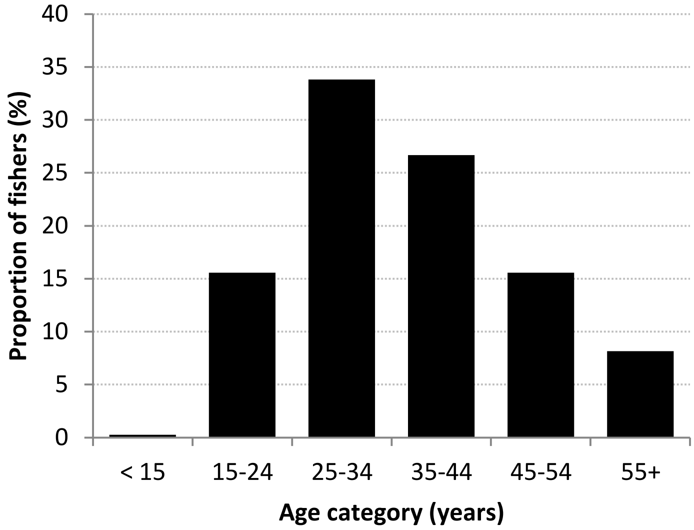


Figure 4.2. Proportion of fishers within each age category. Data source: background interviews (n = 405).

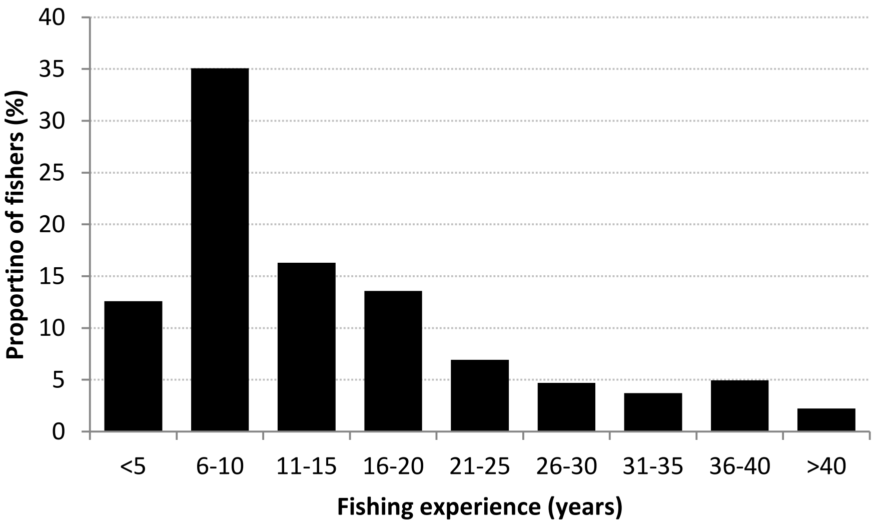


Figure 4.3. Relative frequency of fishers' years of fishing experience. Data source: background interviews (n = 405).

Eighty percent of fishers who participated in background interviews (n = 405) stated that they fished at least once each day, except if they were ill or when unexpected events (such as illness or death of a relative or friend) or extenuating circumstances (such as work in rice field and/or village meetings) arose. Almost 95% of fishers stated that they fished on five or more days each week; those who fished on fewer days nominated work in rice fields and/or the need for rest as reasons for fishing less frequently than others. Most fishers (85%) fished during ten or more months of the year (median 11 months). Catch interview data showed considerable variation in the amount of time fishers spent (a) travelling to their fishing location, (b) fishing after arrival at their location, and (c) per total fishing trip from departure to return (Table 4.1). Mann-Whitney *U* tests confirmed that travel time was significantly greater during the wet season and time fishing after arrival was significantly greater during the dry season; total trip time did not differ significantly between seasons (Table 4.1).

Table 4.1. Median travel, fishing, and total trip durations (hh:mm) estimated by Anororo-based fishers during all catch interviews (n = 1,800), the wet season (n = 528), and the dry season (n = 1,272). Mann-Whitney *U* tests compared fishing times between seasons.

Time	All Median (Range)	Wet season Median (Range)	Dry season Median (Range)	Statistical significance
Time to travel between landing site and fishing location	1:05 (0:01-6:00)	1:10 (0:05-5:00)	1:00 (0:01-6:00)	$p < 0.001$
Time spent fishing after arrival at fishing location	2:00 (0:05-24:00)	1:30 (0:10-11:00)	2:05 (0:05-24:00)	$p = 0.004$
Total trip time (from village departure to village return)	4:00 (0:34-27:00)	4:00 (0:40-19:00)	4:00 (0:34-27:00)	$p = 0.711$

4.3.2 Fishing locations and habitats

Anororo-based fishers use 92 fishing locations (as reported during catch interviews and semi-structured interviews), with individual fishers using a median of 2 (range 1 to 6) fishing locations over the calendar year (scenario interviews; Figure 4.4). Many of the locations are named after adjacent settlements around the lake and marsh. Although some locations are named after current or retired fishers who use or used the location frequently, the names are a way of distinguishing places and do not imply ownership or specific rights; only three locations had two different names, for example, one location was named after the two fishers who were first to fish there. Fishing locations were generally used on a first-come first-served basis and only trap fishers who had built a wall of reeds to attach their traps to had any claim to a fishing territory. Fishing locations were mapped using a GPS and occurred up to 18km north, 10.5km east, 19km south, and 16.4km west of Anororo village (Figure 4.5).

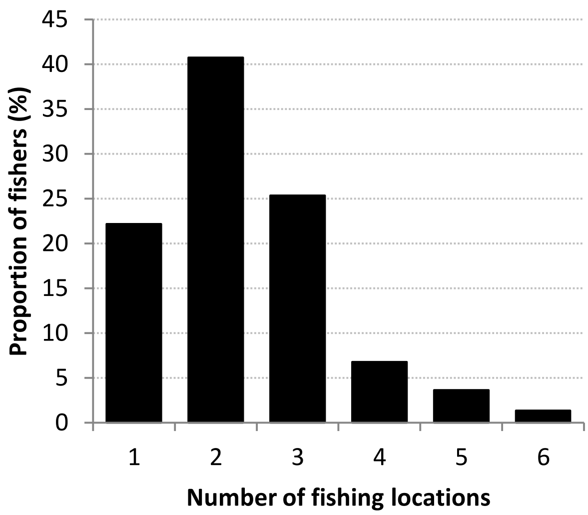


Figure 4.4. Relative frequency of number of fishing locations used by fishers over a calendar year. Data source: scenario interviews (n = 221).

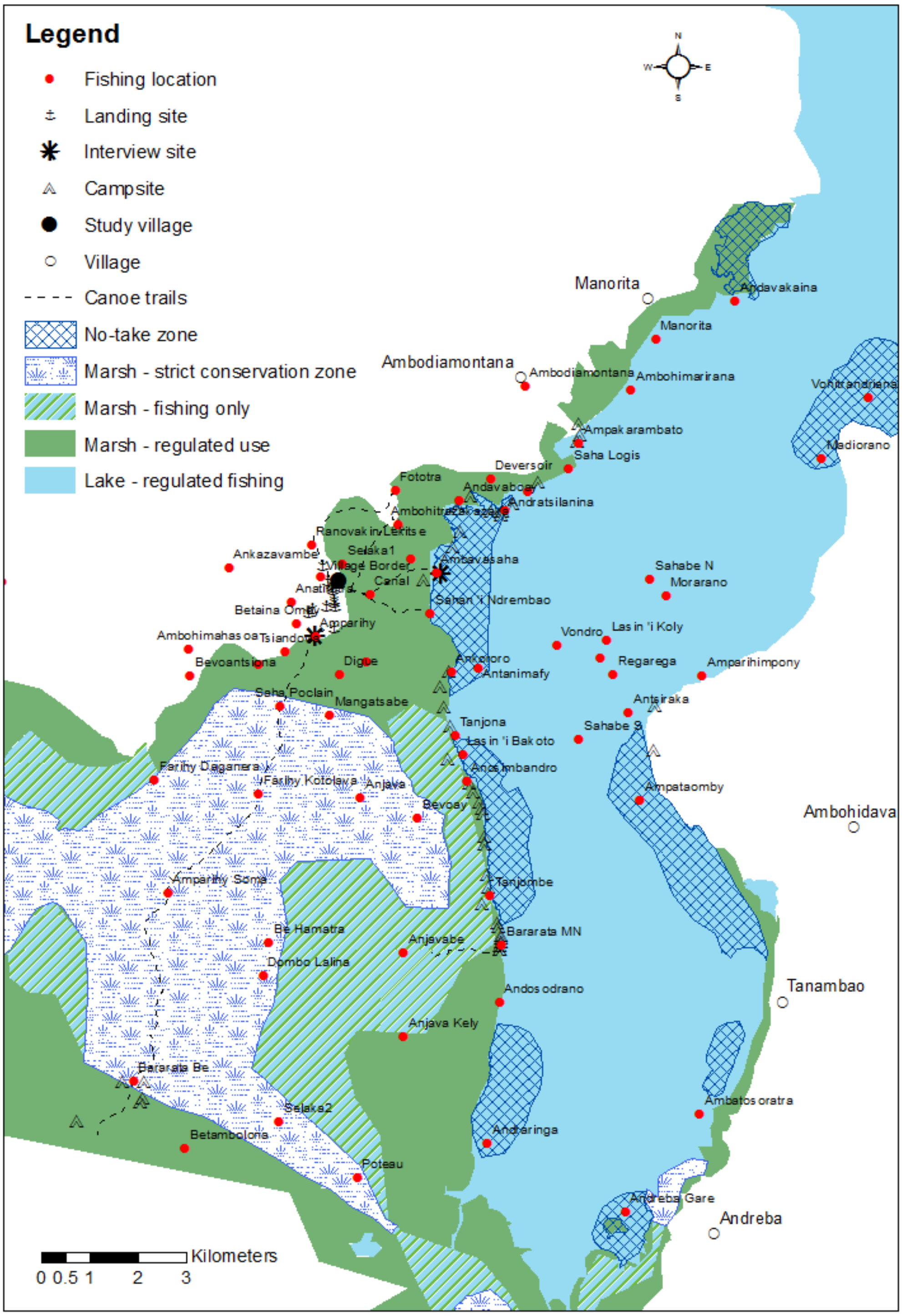


Figure 4.5. Spatial distribution of fishing locations (red dots) in relation to planned management zones within Lake Alaotra and adjacent marsh. Fifteen locations are beyond the area shown on the map; these locations are used infrequently by Anororo fishers and catch interview data were not obtained for them. The canoe trails show the key routes used by fishers to move through the marsh.

The number of fishing locations within each category of habitat varied (Table 4.2). The proportion of fishers using each type of habitat varied across the calendar year (Figure 4.6) and significantly between seasons (Pearson's chi-square test, $\chi^2 = 123.4$, $df = 3$, $p < 0.001$). Frequency of fishing in marsh and at the village border decreases significantly with declining water depth from wet season to dry season; conversely, use of the lake-marsh edge and lake increases significantly from wet to dry season (see Table S4.1 in Appendix S4). Fishers stated that use of fishing locations is greatly influenced by water depth in terms of access, ability to set gear, and presence of fish. There is up to two metres difference in water depth between the high in March-April and the low in November (Figure 4.7). Generally, fishing locations further from the village and closer to the lake are used with greater frequency when water levels are receding. Fishing activity is generally at lowest level in all habitats during December, largely due to the closed period from 15 November to 15 January.

Table 4.2. Number and proportion of fishing locations used by Anororo-based fishers within each type of habitat. Data sources: all interviews (n = 2,426).

Habitat	**n**	**%**
Marsh	45	49
Lake-marsh edge	29	32
Lake	13	14
Village border	5	5
Total	92	100

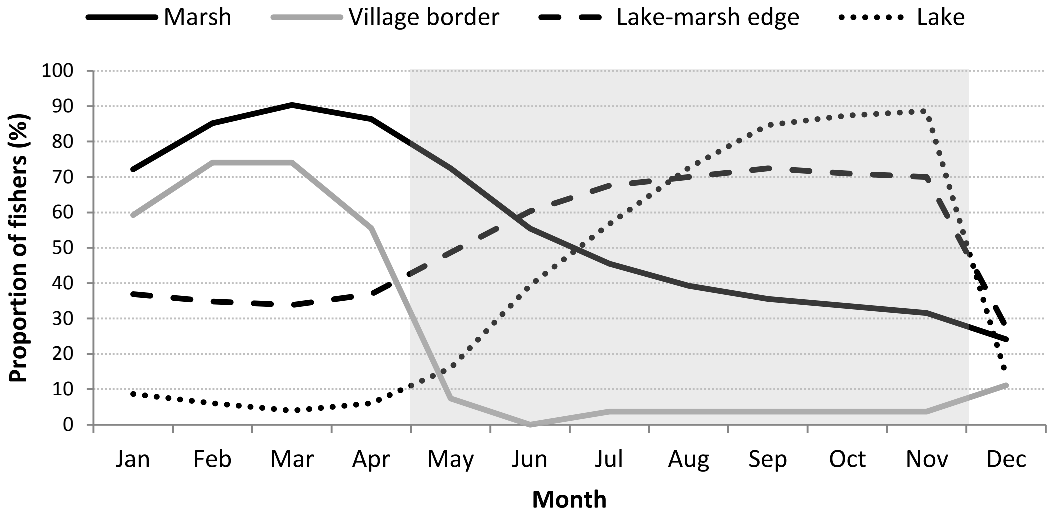


Figure 4.6. Proportions of fishers using each type of habitat across the calendar year. Of fishers fishing within a particular habitat during the year, the proportions refer to the percentage who used that habitat each month. For example, 72% of fishers using marsh habitat fished there during January. The shaded area refers to months of the dry season; unshaded months are in the wet season. Data source: background interviews (n = 405).

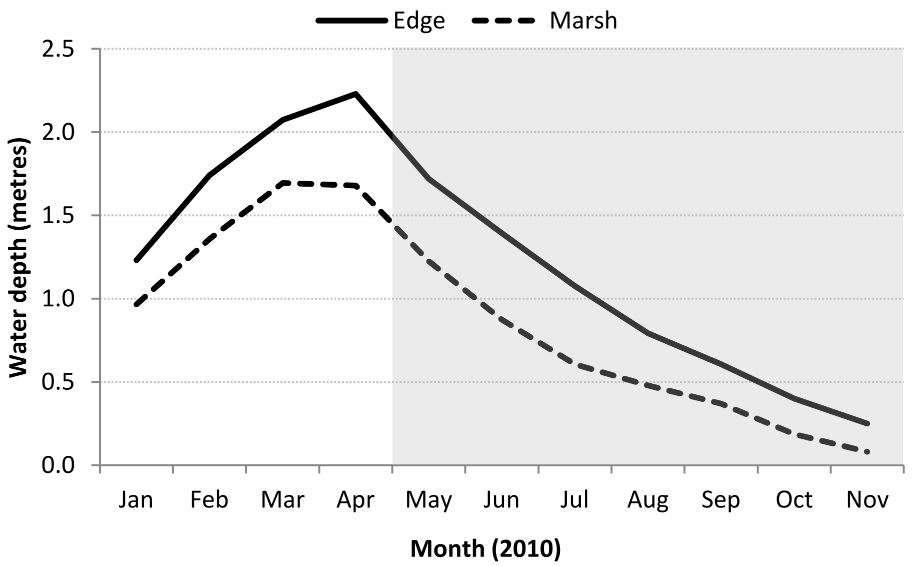


Figure 4.7. Water depth measurements for two sites at Lake Alaotra. The 'Edge' site was located east of Anororo village at the lake-marsh edge and the 'Marsh' site was located in an open area of the marsh approximately 500m south-west of the village. The shaded area refers to months of the dry season; unshaded months are within the wet season.

Water quality parameters did not show unexpected differences across habitats (Table 4.3). Although current water temperature, turbidity, and colour are comparable to those observed by Moreau in 1976, dissolved oxygen content and pH levels have declined, making the wetland more acidic and less favourable for fish.

Table 4.3. Water quality parameters measured across habitats between January and March 2010 (median values are reported; range is shown in parentheses) compared to those for lake habitat in 1976 reported by Moreau (1979a).

Parameter	**Habitat**			**1976**
	Marsh *n = 13 samples across 7 sites*	Lake-marsh edge *n = 13 samples across 6 sites*	Lake *n = 4 samples across 2 sites*	Moreau (1979a)
Water temperature (°C)	24.7 (23.2 to 26.6)	25.2 (23.8 to 26.5)	24.9 (24.2 to 25.9)	20.5 to 28
pH	5.5 (5.3 to 6.0)	5.5 (5.3 to 6.0)	6.0 (n/a)	6.8 to 7.3
Dissolved oxygen (ppm)	3.5 (1.0 to 7.0)	4.0 (2.5 to 7.0)	7.0 (n/a)	1.6 to 21.7
Turbidity (Secchi depth, cm)	35.8 (6.8 to 48.3)	34.5 (24.0 to 51.8)	38.1 (26.0 to 41.0)	25 to 70
Colour of water	Brown-orange to brown-milky	Brown-red to brown-orange	Brown-red	Brown-red

4.3.3 Fishing methods used

Anororo-based fishers use a broad range of methods, and the four most common are traps, gill nets, cast nets, and line & hook; dip nets and two hand methods (rod & bubble and slap & bubble) were also observed (Figure 4.8). Hand methods are used seasonally, typically late in the dry season when water level is lowest. Fishers using these methods usually only fish during the late dry season or use another method for the remainder of the year. My observations indicated that a large number of people (many of whom were not regular fishers) took advantage of low water levels to use hand methods, even though they are prohibited because they target juvenile and/or fish hiding/burrowing in the mud. Some fishing methods reported by fishers were used primarily by fishers outside of Anororo (see Table S4.2).

Figure 4.8. Fishing methods used by Anororo-based fishers. Clockwise from top left: cast net, trap, gill net, dip net, rod & bubble, line & hook.

Traps were consistently used passively overnight with usually 24 hours between fish collections. Gill nets were observed being used in three ways: (i) passively overnight, (ii) passively while waiting, and (iii) actively. In 33% of cases gill net fishers used their nets passively overnight with usually 24 hours between fish collections. In 42% of cases fishers brought their nets with them, set the nets, and then waited for a period of time prior to checking them for fish; this routine of waiting then checking nets for fish was usually repeated twice or several times per fishing trip. This form of passive use of gill nets was observed throughout the year; however, it primarily occurred during the late dry season (Oct-Nov) and early wet season (Jan-Mar) when the risk of damage or loss of gear was greatest. Fishers using gill nets actively (25% of cases) typically had fewer nets and used a baton/club to hit the water and scare fish into the nets; the nets were checked frequently for fish and repositioned within the same fishing location. Cast nets are an active fishing method and were used for a specified number of throws or number of hours. The line & hook method was used passively throughout the day or night; rods were fixed in the mud and then left for a period of time before checking for fish.

Number and dimensions of gear items used per fishing trip varied for each of the four fishing methods most commonly used by Anororo fishers (Table 4.4). Median mesh sizes for traps, gill nets, and cast nets were below the minimum legal size (40mm, 40mm, and 35mm, respectively). Most fishers used mesh sizes smaller than the legal minimum; less than 2% of traps and cast nets used were at or above the minimum legal size and 8% of gill nets had mesh sizes larger than the legal minimum (Figure 4.9). Fishers stated that, over the previous five years, progressively smaller mesh sizes were being used to suit progressively smaller prevailing fish sizes. The mean cost, lifespan, and frequency of repairs for fishing equipment are provided in Table S4.3 in Appendix S4.

Table 4.4. Dimensions of the four fishing methods used most by Anororo-based fishers, as well as number of gear items used per fishing trip. Median values are reported; range is shown in parentheses. Data source: catch interviews (n = 1,800).

Parameter	**Fishing method**			
	Trap	Gill net	Cast net	Line & hook
Length / radius† (m)	0.7 (0.4 to 1.5)	130 (20 to 350)	1.6 (1.1 to 4.0)	n/a
Height / diameter§ (m)	0.5 (0.2 to 1.0)	0.8 (0.2 to 1.5)	n/a	n/a
Area (m^2) / volume§ (m^3)	0.14 (0.01 to 0.9)	100 (5 to 300)	8.0 (3.8 to 50.3)	n/a
Mesh size (mm)	25 (10 to 40)	33 (13 to 75)	18 (8 to 40)	n/a
Number used per trip‡	70 (11 to 310)	7 (1 to 39)	40 (15 to 200)	120 (60 to 300)

† Refers to cast nets.
§ Refers to traps.
‡ Refers to number of throws for cast nets.
n/a = not applicable.

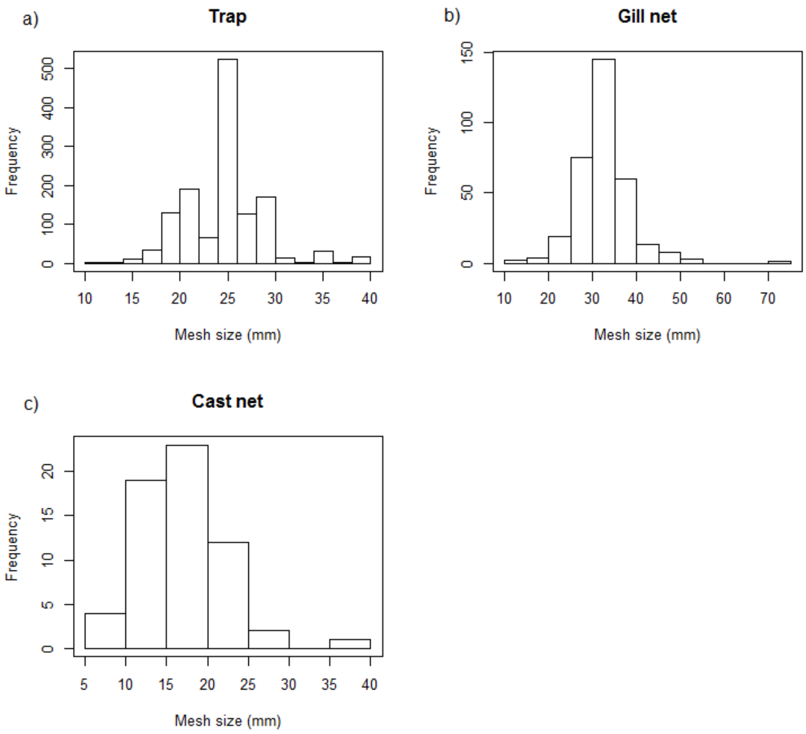


Figure 4.9. Frequency of mesh sizes used for a) traps, b) gill nets, and c) cast nets. Minimum legal size is 40mm, 40mm, and 35mm, respectively. Data source: catch interviews (n = 1,701; traps = 1,310, gill nets = 330, cast nets = 61).

Although some fishers stated during background interviews that they used multiple fishing methods or switched methods across seasons, the majority (78%) stated that they used only one fishing method throughout a year. Traps and gill nets were the predominant fishing methods each season (Figure 4.10); one or both of these methods were used by 94% of fishers during the wet season and by 93% of fishers during the dry season. The range of methods used was not significantly different during the wet season compared to the dry season; however, the extent to which the types of methods were used varied each season (Figure 4.10). As expected, the *jinjira* (rod & bubble) hand method was used significantly more frequently during the dry season.

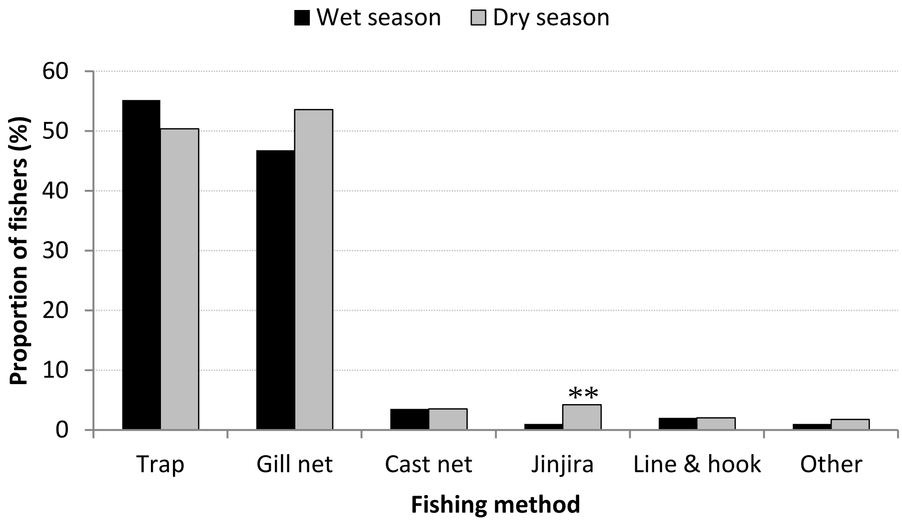


Figure 4.10. Proportion of Anororo fishers using each fishing method during the wet season and the dry season. *Jinjira* is the rod & bubble method. Methods grouped as 'Other' are slap & bubble ('*mangodo*'), dip net ('*sitra*'), and mud enclosure ('*valatany*'). Proportions sum to more than 100% because some fishers (n = 38 wet season; n = 61 dry season) used more than one method. ** Significant difference, $p \leq 0.005$. Data source: background interviews (n = 405).

4.3.4 Species and catch composition

4.3.4.1 Fish species observed

Eleven fish species and a variety of hybrid tilapia were observed in fishers' catches as well as opportunistically outside of formal data collection (Table 4.5; also see Figure S4.1 in Appendix S4). An unidentified species of very small prawn, known locally as *patsa*, was observed in Anororo market during the final phase of data collection. In December 2010 a fisher showed the research team a crayfish (*Procambarus alleni*) he had caught earlier that day, known in Madagascar as *foza orana* because it looks like a cross between a crab, *foza*, and a prawn, *orana*. The species is believed to have been introduced to Madagascar during 2004 or 2005 (Jones et al., 2009) and was observed in southern Lake Alaotra early in 2010; this study was the first to report its presence as far north as Anororo.

Table 4.5. Species of fish observed in Anororo-based fishers' catches or opportunistically during the study.

Common name	Scientific name	Local name	Status	Minimum legal catch length[1]
Nile tilapia	*Oreochromis niloticus niloticus* (Linnaeus, 1758)	Soraka	Introduced 1960	10cm or 13cm
Redbreast tilapia	*Tilapia rendalli* (Boulenger, 1897)	Beloha	Introduced 1955	10cm or 13cm
Mozambique tilapia	*Oreochromis mossambicus* (Peters, 1852)	Malemyloha	Introduced 1960	10cm or 13cm
Paratilapia	*Paratilapia* sp. nov.	Marakely	Native	10cm or 13cm
Hybrid tilapia	*O. niloticus niloticus* and *O. macrochir*	Plakara	-	10cm or 13cm
Hybrid tilapia	Not identified	Kokoloha	-	10cm or 13cm
Hybrid tilapia	Not identified	Lavavava	-	10cm or 13cm
Blotched snakehead	*Channa maculata* (Lacepède, 1801)	Fibata	Invasive; first detected 1980	None
Goldfish	*Carassius auratus auratus* (Linnaeus, 1758)	Trondro gasy	Introduced early 1900s	10cm or 13cm
Common carp	*Cyprinus carpio carpio* (Linnaeus, 1758)	Besisika	Introduced 1926	15cm or 18cm
Black bass	*Micropterus salmoides* (Lacepède, 1802)	Blaky bass	Introduced 1961	None
Indonesian short-finned eel	*Anguilla bicolor bicolor* (McClelland, 1844)	Amalona	Native	8cm or 45cm
Madagascar rainbowfish	*Rheocles alaotrensis* (Pellegrin, 1914)	Ankantrana	Endemic	-
Eastern mosquitofish[2]	*Gambusia holbrooki* (Girard, 1859)	Pirina	Introduced 1940	-

[1] The first minimum legal catch length refers to the dry season (May to November) and the second length refers to the wet season (December to April).
[2] One opportunistic sighting; not observed during catch interviews.

4.3.4.2 Length-weight relationships

Using the equation $W = a \cdot L^b$, where W is weight in grams, L is length in centimetres, and a and b are constants, length-weight relationships were used to estimate fish weight from the length of each fish measured during catch interviews as well as to estimate the total weight of a fishers catch (Table 4.6). The length-weight relationships are comparable to those in the published literature for studies in Madagascar as well as other tropical regions (see Moreau, 1979a; Froese and Pauly, 2010). Figure 4.11 shows length-weight relationships for the two species predominant in fisher catches, Nile tilapia and blotched snakehead.

Table 4.6. Sample size and parameters estimated for length-weight relationships for fish species in Lake Alaotra. Both a and b are constants. R^2 is correlation coefficient.

Species	n	a	b	R^2
Nile tilapia	289	0.0228	2.9170	0.9772
Blotched snakehead	78	0.0044	3.2263	0.9899
Goldfish	64	0.0197	3.0063	0.9070
Common carp	30	0.0371	2.7838	0.9968
Redbreast tilapia	27	0.0106	3.2020	0.8460
Black bass	7	0.0148	2.9961	0.8251
Indonesian short-finned eel*	1	-	-	-
Mozambique tilapia†	1	-	-	-
Unidentified hybrid (Lavavava)†	1	-	-	-
Tilapia (generic)	316	0.0212	2.9425	0.9720

* A length-weight relationship for this species is not available. The relationship for African longfin eel (*Anguilla mossambica*) was used as a proxy to estimate weight (Source: FishBase, *a*=0.0007, *b*=3.2998) (Froese and Pauly, 2010).

† Due to small sample size, lengths and weights for these two fish were not included when calculating a generic length-weight relationship for tilapia.

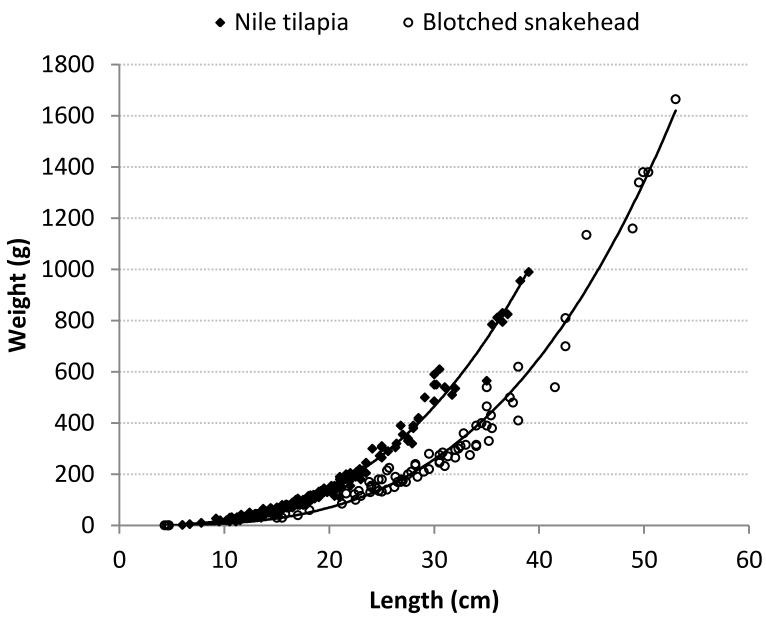


Figure 4.11. Scatterplot of length and weight data showing trendlines for Nile tilapia (*Oreochromis niloticus niloticus*) and blotched snakehead (*Channa maculata*).

4.3.4.3 Catch composition

The species composition of fishers' catches in terms of number and weight of fish caught differed between seasons (Table 4.7). The vast majority (86%) of fish caught were tilapia; although this included several tilapia species, 88% were Nile tilapia and 12% were redbreast tilapia. Nile tilapia and blotched snakehead were predominant in catches, accounting for 86% of fish caught and 87% of catch weight. Due to larger size, snakehead comprised a greater proportion of catch weight (29%) compared to the frequency with which they were caught (11%). This was also the case for common carp and Indonesian short-finned eel; however, these species usually comprised a minor proportion of catches.

Five species (i.e., Nile tilapia, blotched snakehead, redbreast tilapia, goldfish, and common carp) comprised 99.8% of fish within catches and 98.5% of catch weight. The relative proportions of these species in catches differed significantly between seasons (Pearson's chi-square tests, $\chi^2_{(number)} = 116.0$, $\chi^2_{(weight)} = 150.5$, $df = 4$, $p < 0.001$). At species level, Nile tilapia and redbreast tilapia were more prevalent during the dry season than the wet season, whereas blotched snakehead comprised a greater proportion of catches during the wet season than the dry season (Table 4.7).

Moreau (1979a) reported by number and weight the relative proportions of species observed in catches at Lake Alaotra in 1976. At that time two tilapia species, longfin tilapia (*Oreochromis macrochir*; then *Sarotherodon macrochirus*) and redbreast tilapia (*Tilapia rendalli*), accounted for 50% and 18% of catches, respectively, while Nile tilapia, *O. niloticus niloticus* (then *S. niloticus*) accounted for only 7%. Other striking differences in the 1976 proportions compared to 2009/2010 are the prevalence of common carp at 19% (now only 1.1%) and black bass at 5% (now <1%), as well as the absence of blotched snakehead (first observed in Lake Alaotra in 1980). One similarity between catch compositions over time is the high proportion of all tilapia species combined, amounting to 75% in 1976 and 86% in 2009/2010.

Table 4.7. Species composition of fisher's catches by number of fish caught and by catch weight across the study period. Values in bold are significantly different across seasons (one-sample chi-square tests). Proportions by number and weight for species in 1976 are also shown. [a]Plakara (longfin tilapia *Oreochromis macrochir*) were not hybrid in 1976. Data source: catch interviews (n = 1,767) and Moreau (1979b).

Species	BOTH SEASONS				WET SEASON				DRY SEASON				*p*	1976	
	Number		Weight		Number		Weight		Number		Weight			Number	Weight
	n	%	kg	%	n	%	kg	%	n	%	kg	%		%	%
Nile tilapia	48,108	75.33	2,807.38	58.55	10,773	**65.52**	651.74	**45.80**	37,335	**78.74**	2,155.64	**63.93**	0.001	7	7
Redbreast tilapia	6,488	10.16	216.35	4.51	1,159	**7.05**	30.41	**2.14**	5,329	**11.24**	185.95	**5.51**	0.002	18	16
Mozambique tilapia	15	0.02	0.64	0.01	7	0.04	0.25	0.02	8	0.02	0.38	0.01		+	+
Paratilapia	1	< 0.01	0.04	< 0.01	0	0	0	0	1	< 0.01	0.04	< 0.01			
Hybrid tilapia – Plakara	89	0.14	4.35	0.09	2	0.01	0.09	0.01	87	0.18	4.26	0.13		50[a]	50[a]
Hybrid tilapia – Kokoloha	2	< 0.01	0.12	< 0.01	0	0	0	0	2	< 0.01	0.12	< 0.01			
Hybrid tilapia – Lavavava	3	< 0.01	0.15	< 0.01	1	0.01	0.04	< 0.01	2	< 0.01	0.11	< 0.01			
All tilapia	54,706	85.67	3,029.03	63.17	11,942	72.63	682.53	47.97	42,764	90.19	2,346.5	69.59		75	73
Blotched snakehead	6,951	10.88	1,366.84	28.51	3,824	**23.26**	662.69	**46.57**	3,127	**6.59**	704.15	**20.88**	<0.001	n/a	n/a
Goldfish	1,450	2.27	131.53	2.74	427	2.60	27.84	1.96	1,023	2.16	103.69	3.08			
Common carp	714	1.12	202.01	4.21	231	1.40	39.72	**2.79**	483	1.02	162.29	**4.81**	0.022	19	21
Black bass	6	0.01	0.89	0.02	6	0.04	0.89	0.06	0	0	0	0		5	4
Indonesian short-finned eel	33	0.05	14.97	0.31	13	0.08	9.39	0.66	20	0.04	5.58	0.17			
Madagascar rainbowfish	n/a	n/a	49.50	1.03	n/a	n/a	0	0	n/a	n/a	49.50	1.47			
Total	63,860	100	4,794.77	100	16,443	100	1,423.06	100	47,417	100	3,371.71	100			

Catch composition differed significantly across habitats in terms of number of fish caught (Pearson's chi-square test, $\chi^2 = 353.6$, $df = 18$, $p < 0.001$) and weight of fish caught (Pearson's chi-square test, $\chi^2 = 518.9$, $df = 21$, $p < 0.001$). In both seasons there was a significantly greater proportion of snakehead in catches from the marsh compared to other habitats; in the wet season there were significantly greater proportions of goldfish and carp in catches from the lake (Figures 4.12 and 4.13). Catches from the lake comprised significantly greater proportions of Nile tilapia during the dry season than the wet season in terms of both number of fish caught (one-sample chi-square test, $\chi^2 = 312.0$, $df = 1$, $p < 0.001$) and weight of fish caught (one-sample chi-square test, $\chi^2 = 332.0$, $df = 1$, $p < 0.001$). By weight, snakehead comprised the majority of catches in the marsh during the wet season, whereas in the dry season tilapia was predominant in each habitat.

Across all habitats, the results in Figures 4.12 and 4.13 (and Table 4.7) show that catch composition differs markedly depending on whether it is expressed in terms of number or weight of fish caught, particularly in the wet season. Seasonal differences in catch compositions were probably primarily due to shifts in fishers' use of habitat over the year, from marsh and village border locations in the wet season to lake and lake-marsh edge locations during the dry season (see Section 4.3.2). For example, increased use of lake and lake-marsh edge during the dry season made it more likely to catch Nile tilapia and less likely to catch blotched snakehead due to differences in relative abundance within these habitats at this time.

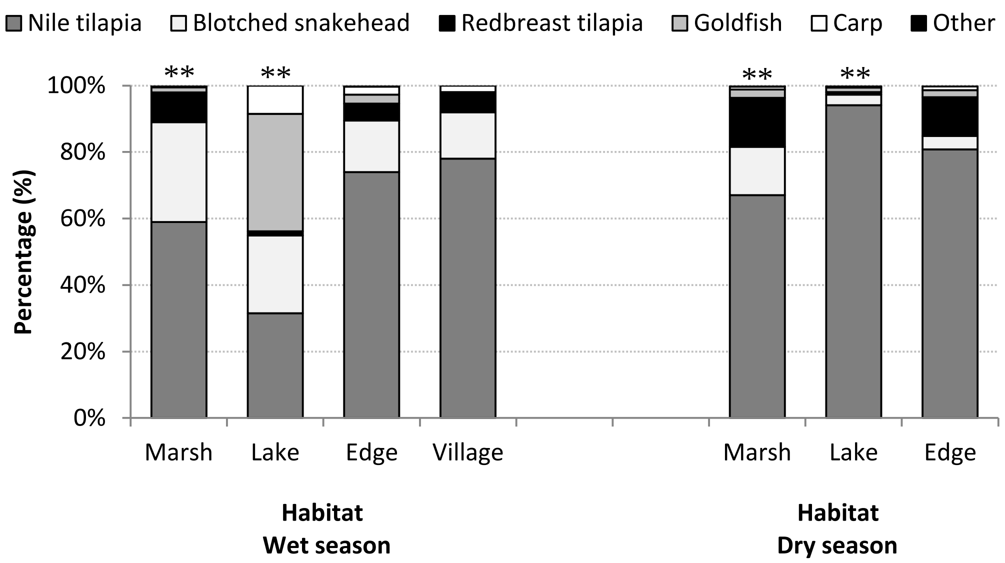


Figure 4.12. Proportion of catch based on number of fish for each habitat and season. Species grouped as 'Other' are Mozambique tilapia, Paratilapia, hybrid tilapia, black bass, and Indonesian shorted-finned eel. Chi-square tests confirmed significant differences in catch composition across habitats. ** p < 0.001. Data source: catch interviews (n = 1,761).

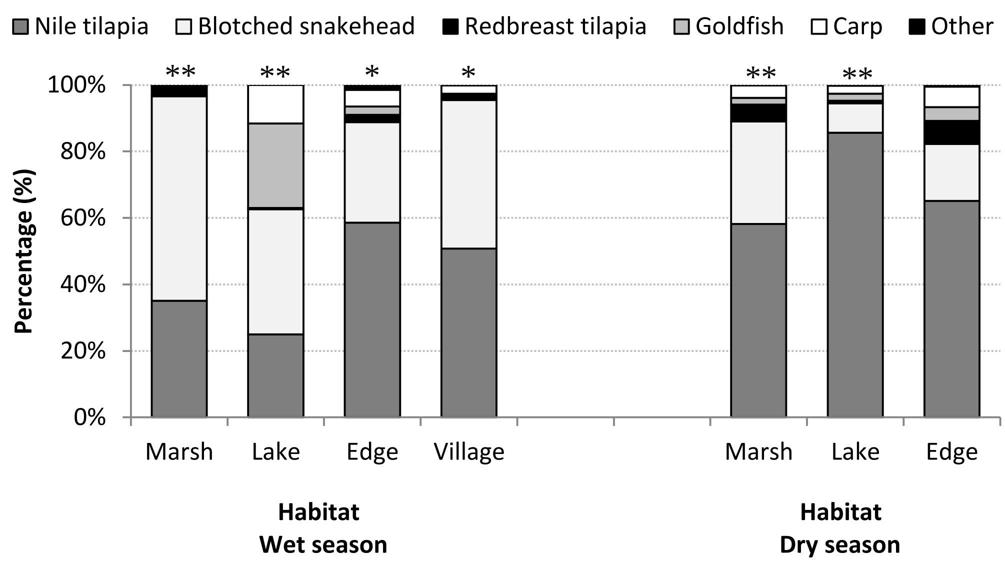


Figure 4.13. Proportion of catch based on weight of fish for each habitat and season. Species grouped as 'Other' are Mozambique tilapia, Paratilapia, hybrid tilapia, black bass, and Indonesian shorted-finned eel. Chi-square tests confirmed significant differences in catch composition. * p < 0.01; ** p < 0.001. Data source: catch interviews (n = 1,761).

The fishing methods used by 91% of Anororo-based fishers were traps, gill nets, or cast nets, and catch composition varied according to the method used. Nile tilapia and blotched snakehead were predominant in catches using traps (Figure 4.14a); gill net catches were predominantly Nile tilapia (Figure 4.14b). While catches using cast nets were also mostly Nile tilapia, they included a greater proportion of redbreast tilapia compared to other methods (Figure 4.14c). Cast net catches also included a greater proportion of carp by weight, because carp caught in cast nets are heavier (and hence attract a higher sale price) than those caught with other methods. Limited space to use gill nets or cast nets in marsh habitats meant that traps, which need to be fixed to vegetation, were predetermined to catch more blotched snakehead. The line & hook method (not shown) uses live bait and therefore targets the carnivorous blotched snakehead, which comprised a median of 92% of catches using this method.

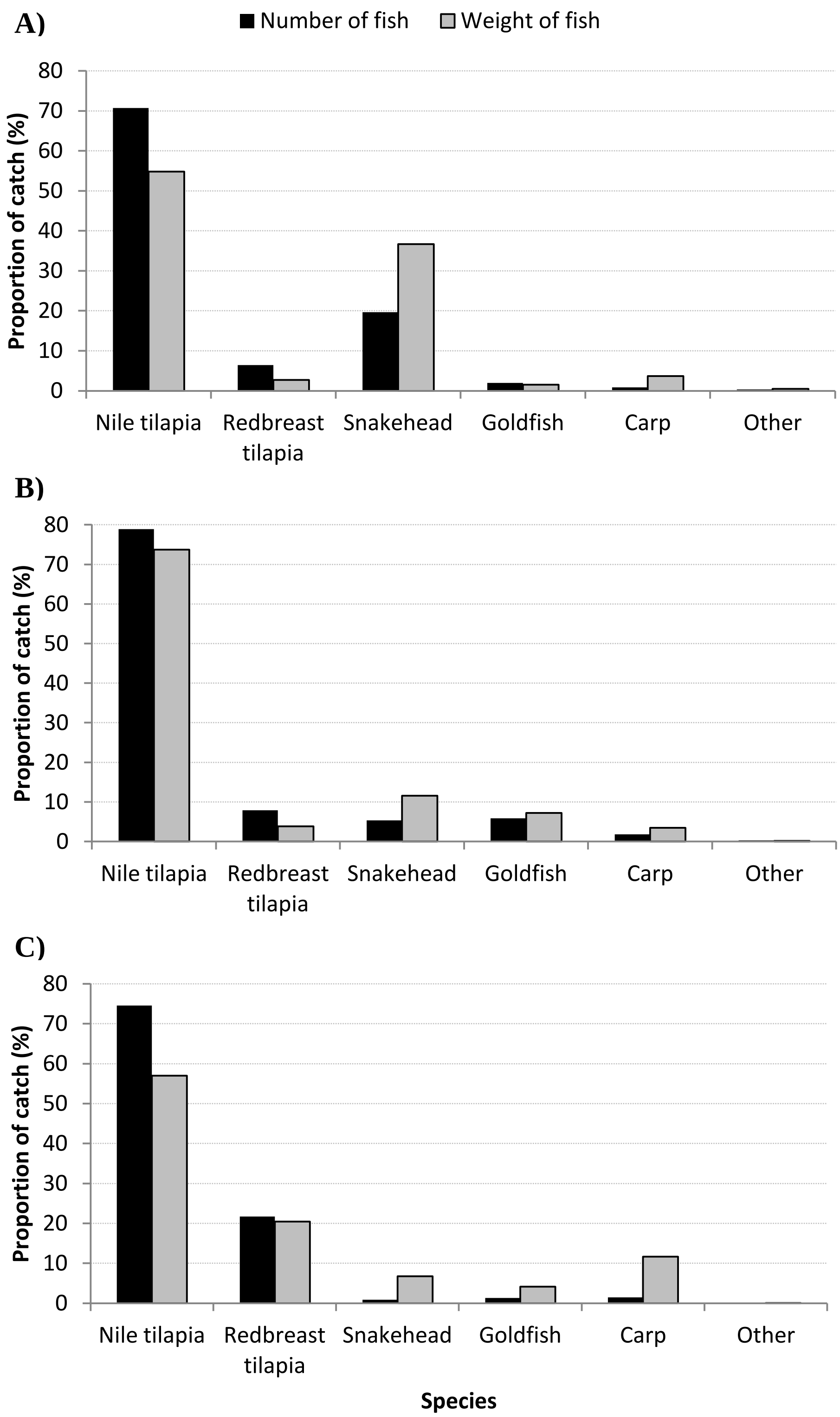


Figure 4.14. Proportion of fish species present in catches using A) traps (n = 30,094 fish), B) gill nets (n = 10,670 fish), and C) cast nets (n = 17,073 fish). Proportions refer to number of fish in catch and weight of fish in catch. Species grouped as 'Other' are Mozambique tilapia, Paratilapia, hybrid tilapia, black bass, and Indonesian short-finned eel.

4.3.4.4 *Temporal variation in fish lengths and weights*

The weights of fish caught varied across seasons (Table 4.8). Median weight of Nile tilapia was greater during the wet season than the dry season. However, this result is attributable to the disproportionately large number of juvenile Nile tilapia caught during the dry season, primarily by cast net fishers. Fifty-three percent of Nile tilapia caught during the dry season were less than 13cm in length, compared to 42% during the wet season. Nile tilapia caught during the dry season were larger than during the wet season if cast net catches are excluded from analysis. Median weights for the other four species caught most frequently were greater during the dry season irrespective of fishing method used. These patterns occurred because most fish species in Lake Alaotra breed once per year, when water level begins to rise in October or November, and then grow through the year. Median weights for most fish species were significantly greater in 2009 compared to 2010 (Table 4.9). Fish growth in floodplain fisheries appears to be strongly affected by flood extent, with strong positive impacts of flood on fish growth (Halls and Welcomme, 2004; Halls et al., 2011). This is due to both density-dependent effects on growth, with slower growth at high densities (such as low water levels) (Lorenzen, 1996), and increased productivity in wetter years (Kolding and van Zwieten, 2012). In this study 2009-2010 was a drier year than 2008-2009 (based on TAMSAT data; Grimes et al., 1999), which is likely to have had a negative impact on fish growth.

Table 4.8. Median weights (in grams) of fish species each season in 2010; range is shown in parentheses. Mann-Whitney *U* tests confirmed significant differences in weights between seasons. Data source: catch interviews (n = 1,418).

Species	**n**	**Wet season**	**n**	**Dry season**	**Statistical significance**
		Weight (g)		Weight (g)	
Nile tilapia	10,773	44.2 (1.0 to 822.5)	24,593	36.1 (1.1 to 997.9)	$p < 0.001$
Blotched snakehead	3,824	133.5 (5.3 to 2,212.2)	1,863	161.7 (10.4 to 2,323.9)	$p < 0.001$
Redbreast tilapia	1,159	23.6 (3.3 to 155.3)	3,886	25.0 (1.8 to 317.3)	$p < 0.001$
Common carp	231	83.4 (20.1 to 1,675.6)	434	185.1 (22.6 to 2,220.2)	$p < 0.001$
Goldfish	427	56.2 (10.2 to 395.9)	822	102.1 (17.1 to 605.6)	$p < 0.001$

Table 4.9. Median weights (in grams) of fish species across years; range is shown in parentheses. Analysis compared catches in June, July, October, and November each year because these were the only months where data were available for both years. With the exception of goldfish, Mann-Whitney *U* tests confirmed significant differences in weights between years. Data source: catch interviews (n = 824).

Species	**n**	**2009** Weight (g)	**n**	**2010** Weight (g)	**Statistical significance**
Nile tilapia	12,742	61.5 (2.5 to 855.8)	14,630	36.1 (1.1 to 997.9)	$p < 0.001$
Blotched snakehead	1,264	171.9 (16.8 to 1,376.5)	1,045	151.8 (15.2 to 2,323.9)	$p < 0.001$
Redbreast tilapia	1,443	34.5 (3.3 to 173.4)	1,211	27.1 (5.4 to 317.3)	$p < 0.001$
Common carp	49	652.8 (52.0 to 1,746.0)	147	255.0 (22.6 to 2,220.2)	$p = 0.001$
Goldfish	201	70.4 (14.6 to 400.3)	421	96.8 (20.6 to 250.9)	$p = 0.1$

4.3.4.5 *Length-frequency distribution*

Length-frequency data for Nile tilapia and blotched snakehead are illustrated in Figure 4.15. Ninety percent of Nile tilapia caught during the study were less than length at first maturity (Lm); 50% of blotched snakehead caught were less than Lm (see Froese and Pauly, 2010).

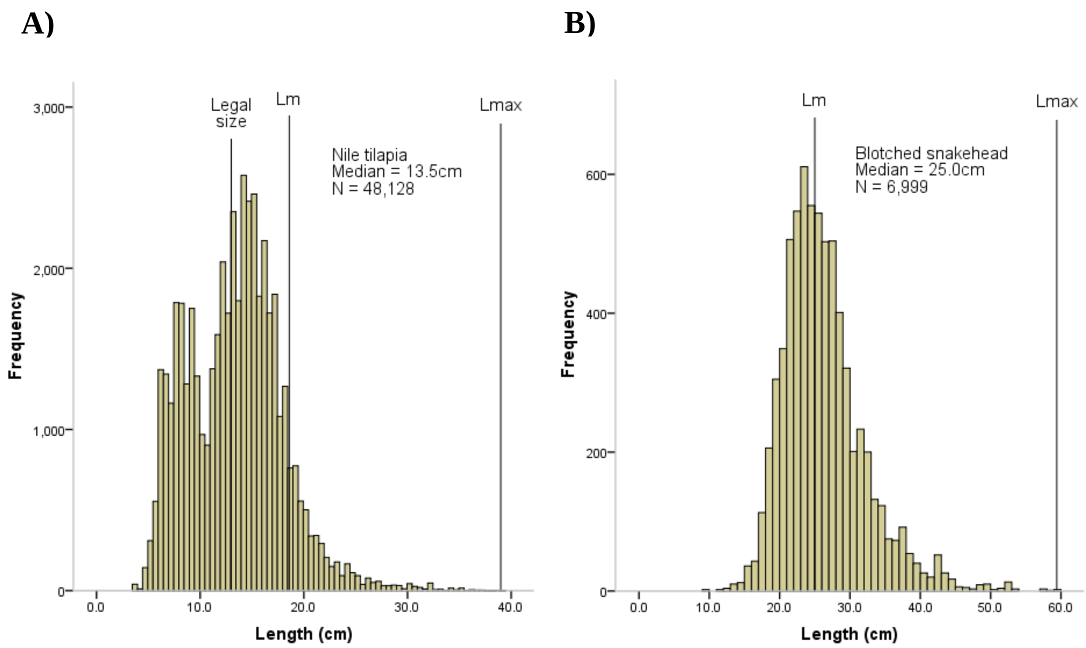


Figure 4.15. Length-frequency distributions for (A) Nile tilapia and (B) blotched snakehead; mean length at first maturity (Lm) is 18.6cm and 25cm, respectively (source: www.fishbase.org). Lmax indicates the maximum length of fish caught during the study (39cm and 59.4cm, respectively). Legal size refers to the minimum permissible length of Nile tilapia in catches (13cm); there is no minimum legal size for blotched snakehead. Data source: catch interviews (n = 1,761).

4.3.4.6 *Timing of fish reproduction*

The current timing of the annual fishery closure (15 November to 15 January) is intended to provide respite for spawning fish, particularly for carp that only reproduce once each year in Lake Alaotra. Fish eggs and/or fry were observed in catches during 38 catch interviews conducted between 21 October and 10 November 2009 and between 27 September and 12 November 2010. These were in catches across gear types and from 12 fishing locations; the majority of locations (58%) were in lake-marsh edge habitat. Nile tilapia, common carp, and goldfish were observed with eggs as early as September, and by mid-November no eggs or fry were observed in catches; eggs and fry were not observed at any other time of year. These observations are supported by Moreau's findings on spawning by other tilapia species in Lake Alaotra in the 1970s (Moreau, 1982). Owners of local restaurants also noted that many carp and tilapia they bought during October were gravid, and suggested that the fishery closure occurs too late. Similarly, 34% of 221 fishers completing scenario interviews stated that the timing of the closed period should be brought forward to provide better protection for spawning fish and increase young-of-year survival; this would also make it possible to re-open the fishery in time for Christmas and New Year celebrations. Timing of the fishery closure and fisher suggestions for management are explored further in Chapter 7.

4.3.5 Catch size and income from fishing

Although values varied between fishing methods, across all methods and both seasons fishers considered catch sizes of fewer than 15 average-sized fish (approximately 15cm in length) or less than 1kg to be 'bad' days. Fishers defined 'average' and 'good' days as catching a median of 40 fish (approximately 2.5kg) and 100 fish (approximately 6kg), respectively (Table 4.10). These results are consistent with definitions of good, average, and bad fishing days from focus group sessions exploring general characteristics of the fishery.

Table 4.10. Median number of fish caught on good, average, and bad fishing days for each fishing method across seasons, as defined by fishers during background interviews (n = 405).

Fishing method	Both seasons			Wet season			Dry season		
	Good	Average	Bad	Good	Average	Bad	Good	Average	Bad
Trap	80	40	10	80	40	10	80	40	12.5
Gill net	120	40	20	80	40	13	120	40	20
Cast net	160	60	20	120	57	20	160	80	35
Line & hook	80	35	15	80	28	5	80	40	20
Rod & bubble	50	26	15	n/a	n/a	n/a	50	20	15
Slap & bubble	100	40	20	n/a	n/a	n/a	120	40	22
All methods	**100**	**40**	**15**	80	40	10	120	40	20

n/a = not applicable

The majority of catches observed during catch interviews, both in terms of number of fish caught and total catch weight, were either average or bad days as defined by fishers (Figure 4.16). For 13 (0.7%) fishing trips fishers did not catch any fish; 15% of catches comprised five or fewer fish, and 30% of catches comprised fewer than 10 fish. Median catch size was 16 fish, ranging from 0 to 752 fish. Median catch weight was 1.61kg, ranging from 0 to 46.9kg.

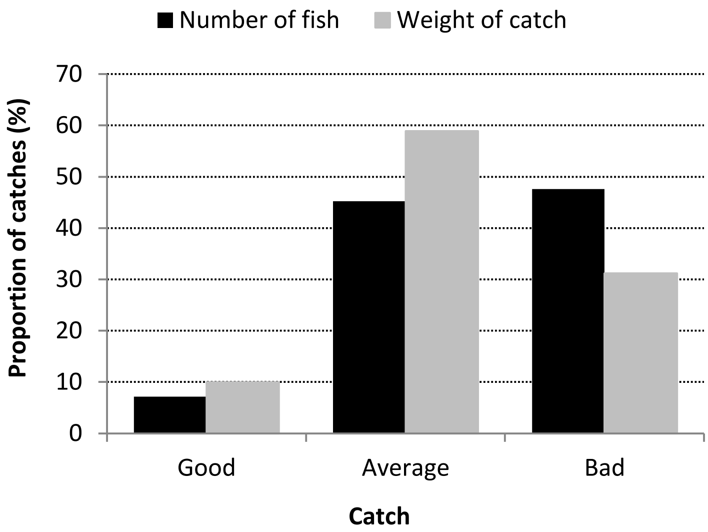


Figure 4.16. Relative frequency of catches for good, average, and bad days, as defined by fishers, in terms of number of fish caught and weight of catch. Good = 100 fish or >6kg, Average = 40 fish or 2.5kg, Bad = <16 fish or 1kg. Data source: catch interviews (n = 1,761).

Total catch weight varied by month over the study period (Figure 4.17). Number of fish caught and total catch weight differed significantly between months (Kruskal-Wallis tests: $\chi^2_{(number)} = 110.3$, $\chi^2_{(weight)} = 129.7$, $df = 14$, $p \leq 0.001$) as well as between seasons during 2010 (Mann-Whitney U tests: $n_{(wet)} = 523$ $n_{(dry)} = 895$, $U_{(number)} = 217057$, $p < 0.022$; $U_{(weight)} = 205766.5$, $p < 0.001$).

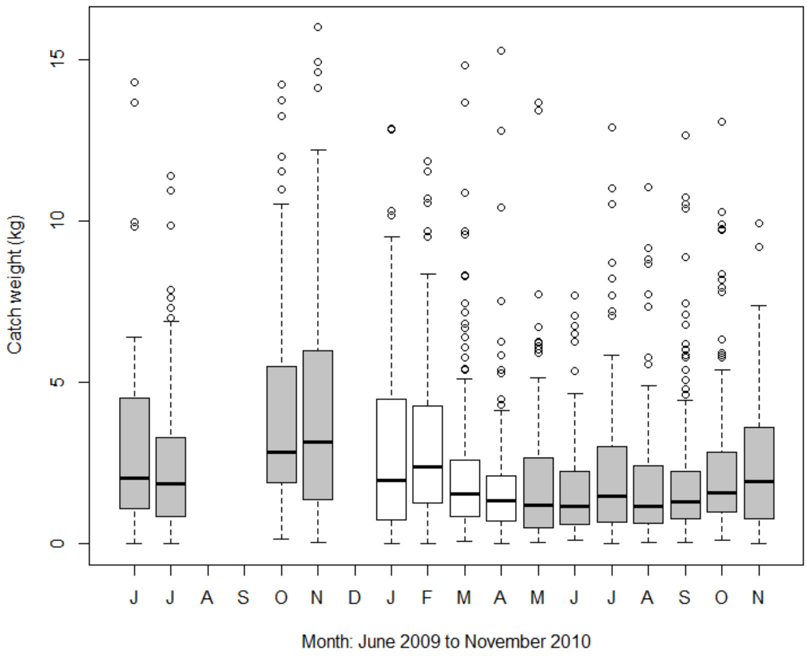


Figure 4.17. Box and whisker plot of catch weight (kg) by month over the study period. No sampling occurred in August, September, or December in 2009. The horizontal bar represents the 50th percentile, the top of the box the 75th percentile, and the base of the box the 25th percentile. Whiskers represent the range of data, and open circles are outliers; not all outliers are shown. Grey shading refers to months of the dry season. Data source: catch interviews (n = 1,761).

A median of 90% of fish within a catch were sold; the remaining 10% were kept for food. The proportion of fish sold increased slightly with catch size ($r = 0.224$, $p < 0.0001$). Fishers sold their entire catch in 39% of cases where fish were caught (n = 1,748); in 22% of cases fishers kept their entire catch for food. Instances of fishers selling or keeping their entire catch were observed each season. In 2010 fishers sold a median of 94% of fish caught during the dry season and 82% of fish caught during the wet season; these proportions were significantly different (Mann-Whitney *U* test: $n_{(wet)} = 520$ $n_{(dry)} = 889$, $U = 270682.5$, $p < 0.001$).

Whole or part catches are purchased from fishers by collectors. Collectors estimate the value of the catch by eye, and value depends on the quantity (fullness of a 15L bucket) and size of fish. Although fishers often claimed that fish prices are unstable, fish collectors stated that fish prices did not vary over the year and that they kept to a consistent set of decision criteria when estimating catch value. Although carp was valued highest, followed by tilapia and then snakehead, a large snakehead was worth more than a small carp or small tilapia. Because catches typically comprised a mix of species, prices were primarily determined by fish size rather than species. Generally, a bucket of small fish sold for 12,000 Ariary, medium fish (approximately 15cm in length) for 13,000 Ariary, and large fish (approximately 20cm in length; caught with mesh ≥32mm) for 15,000 Ariary. Catches also usually included a mix of fish sizes, and catch prices varied according to the mix and type of collector (Figure 4.18). The colour of fish was also stated to be a criterion of value. Darker fish from the marsh were perceived to taste better and obtain higher prices, although there was no empirical evidence to confirm this.

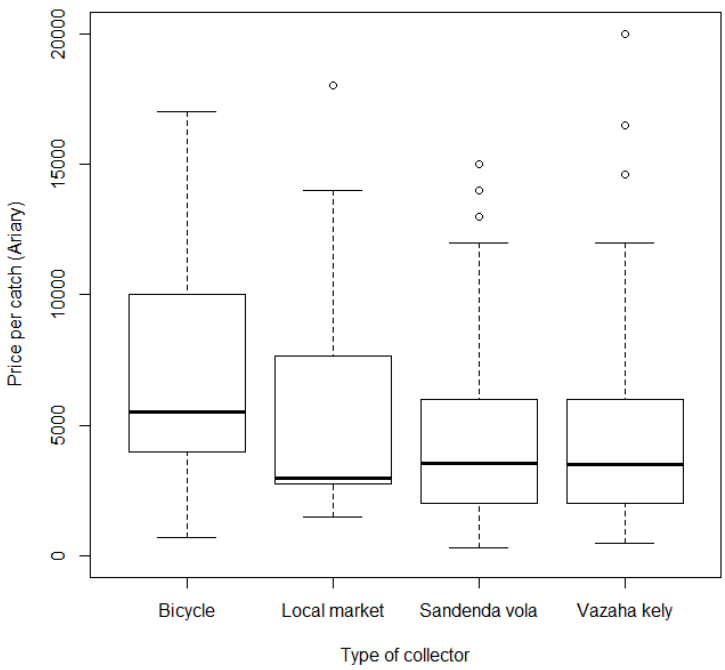


Figure 4.18. Box and whisker plot of catch prices by type of collector. The horizontal bar represents the 50th percentile, the top of the box the 75th percentile, and the base of the box the 25th percentile. Whiskers represent the range of data, and open circles are outliers; 3 outliers are not shown. Data source: catch interviews (n = 258).

Income from sale of catches varied between fishing methods used as well as seasons. The median price received per kilogram of catch was 1,763Ar (or 170Ar per fish sold) across all fishing methods. However, the range of income received per catch for each fishing method was considerable: for example, 443 to 9,941Ar/kg for traps and 890 to 6,570Ar/kg for gill nets (Table 4.11).

Table 4.11. Median price received in Ariary* per kilogram and per fish for fish sold by fishers using each fishing method. Data source: catch interviews (n = 258).

Fishing method	Both seasons			Wet season			Dry season		
	n	Price per kg	Price per fish	n	Price per kg	Price per fish	n	Price per kg	Price per fish
Trap	158	1,735	200	53	1,403	200	105	2,021	191
Gill net	64	1,896	145	9	1,690	135	55	1,958	150
Cast net	6	1,283	29	1	1,460	23	5	1,197	34
Line & hook	8	1,413	500	3	1,264	208	5	1,785	857
Rod & bubble	9	1,668	107	0	n/a	n/a	9	1,668	107
Slap & bubble	13	1,781	26	0	n/a	n/a	13	1,781	26

* Malagasy Ariary (Ar). At the time of the study GBP£1 = 3,000Ar and USD$1 = 2,080Ar.
n/a = not applicable.

Results from the general linear model indicated that season, total weight of fish sold, and the type of collector were significant predictors of the price fishers received for their catch; gear type, habitat type, and fish species were not significant explanatory variables (Table 4.12). These results confirm that the price of fishers' catches depended on the total weight of catch sold as well as the type of collector sold to. Bicycle collectors typically provided higher prices and worked very early each morning to allow time to transport fish to other village markets. These findings are consistent with Copsey et al. (2009b), noting that bicycle collectors offered premium prices for early morning sales and the freshest fish. In contrast, commercial collectors worked until approximately midday, when they returned to the village to resell fish to commercial agents with trucks. Local market collectors worked at all times of the day and predominantly purchased small fish for sale in the Anororo market.

Table 4.12. Results of the linear model of factors explaining price received by fishers. [a, b, c, d] Baseline levels are 'cast net', 'edge habitat', 'dry season', 'bicycle collector', respectively. Significant values are in bold.

Explanatory variables	Estimate	SE	*t*	*P*
Intercept	1.946497	0.282416	6.892	<0.0001
Gear type[a]				
Rob & bubble	0.004289	0.210432	0.020	0.9838
Line & hook	0.416547	0.222195	1.875	0.0621
Slap & bubble	-0.101995	0.185849	-0.549	0.5837
Gill net	0.308034	0.170163	1.810	0.0715
Trap	0.267393	0.167133	1.600	0.1109
Habitat[b]				
Lake	-0.134897	0.096509	-1.398	0.1635
Marsh	0.050405	0.068055	0.741	0.4596
Season[c]				
Wet	-0.269232	0.063145	-4.264	**<0.0001**
Weight of tilapia	0.017002	0.013323	1.276	0.2032
Weight of snakehead	-0.022622	0.009225	-2.452	**0.0149**
Weight of carp	0.013730	0.014160	0.970	0.3332
Weight of goldfish	0.009882	0.011013	0.897	0.3705
Weight of other species	0.071724	0.042106	1.703	0.0898
Total weight of fish sold	0.814416	0.030453	26.743	**<0.0001**
Collector[d]				
Local market	-0.160528	0.136086	-1.180	0.2393
Sandenda vola	-0.207358	0.074692	-2.776	**0.0059**
Vazaha kely	-0.237183	0.070839	-3.348	**0.0009**

Multiple R-squared = 0.836 Adjusted R-squared = 0.8243
F-statistic = 71.66 on 17 and 239 DF p-value <0.0001

4.3.6 Fishers' perceptions of the state of the fishery

During background interviews, the majority of fishers stated that catch sizes and fish sizes had been decreasing each year from 2005 to 2009 (Figure 4.19). Of the fishers who perceived a decrease in catch size (n = 316), 32% stated the reason was increased numbers of fishers. Other frequently stated reasons were use of destructive fishing methods (19%; n = 61), invasive aquatic plants clogging and degrading fishing locations (18%; n = 58), poor fish reproduction (15%; n = 47), and declining or low water levels (13%; n = 40). Thirty percent (n = 70) of fishers who perceived a decrease in fish size (n = 234) stated lack of food for fish and general degradation of the marsh as the primary reason. Other reasons included (i) stressing of fish due to excessive disturbance (19%; n = 45), (ii) overfishing (all the large fish have been caught and only small fish remain, 21%; n = 48), and (iii) poor water quality caused by invasive aquatic plants (15%; n = 34). Of the 221 fishers interviewed in 2010, 91% stated that fishing over the previous year had been bad and described the Alaotra fishery as in decline, deteriorating, or not profitable. The most frequently stated reasons for decline were low water level (42%; n = 84), fishers using destructive methods that targeted juvenile fish (18%; n = 37), decreasing fish stock (17%; n = 35), and climate change (i.e., a long and cold winter, 12%; n = 24). However, fishers also referred to a Malagasy adage regarding the number of fish in the lake and the quantity fishers catch. *"Raha omen'ny Tanororo haza"* translates literally as "*If the Tanororo gives you catch"* and relates to the spirit owning the water. Fishers do not take it for granted that they will catch fish and to some extent believe they do not have full control over their catch; the water has 'owners' and catch depends on whether the 'owners' give it to you.

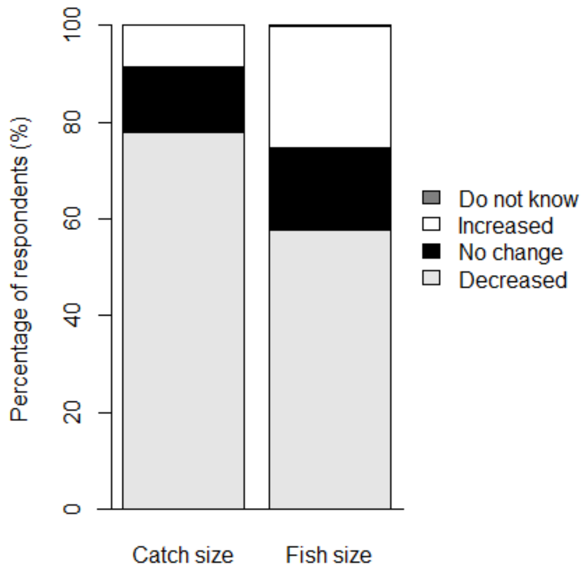


Figure 4.19. Fishers' perceptions of the state of fish resources in Lake Alaotra (n = 405). The responses "do not know", "increased", "no change", and "decreased" refer to perceptions of how catch sizes and fish sizes have changed over time.

Although 344 (85%) of 405 fishers interviewed stated they wanted to continue fishing, 330 (81%) did not want their son to become a fisher. When asked why, 200 of these fishers stated they wanted their son to continue education and get a better job. Other fishers stated that the fishery was in decline or no longer profitable, or that being a fisher was difficult and tiring work. The 73 fishers (18%) who wanted their son to become a fisher stated as reasons no other feasible option (e.g., no land for rice cultivation, no other jobs), family tradition (like father, like son), or to work with or help his father.

4.4 DISCUSSION

The Lake Alaotra wetland is of considerable importance for biodiversity and as a source of food and/or income for more than half a million people (Ramanampamonjy et al., 2003; Andrianandrasana et al., 2005). The wetland supports a variety of endemic and endangered species including the Alaotran gentle lemur (*Hapalemur alaotrensis*), Madagascar pochard (*Aythya innotata*), Meller's duck (*Anas melleri*), and Madagascar rainbowfish (*Rheocles alaotrensis*), and sustains an artisanal fishery of high value to local and regional economies (Ramanampamonjy et al., 2003). Unmanaged or excessive fishing activity in inland fisheries such as Lake Alaotra could adversely impact adjacent wetland habitat, biodiversity, and ecosystem integrity, and also lead to dramatic changes in the size structure of targeted fish species, shifts in catch composition and overall fish assemblage, declining returns for fishing effort, and livelihood insecurity (Hilborn et al., 2003; Eggert and Lokina, 2010; Welcomme et al., 2010). This study is the first comprehensive investigation of the functioning and status of the Alaotra fishery in almost four decades (see Moreau, 1979a, b). Because the study characterises the fishery in terms of where fishers fish, the gear they use, what they catch, and the effort involved, it is a fundamental step towards effective future management of the fishery (see Holland and Herrera, 2009; Welcomme, 2011a).

With multiple habitats, gear types used, and targeted species as well as varying fisher effort and catch sizes, Lake Alaotra's fishery is similar in structure to many other small-scale artisanal fisheries in developing countries (see Balirwa et al., 2003; Lorenzen et al., 2006; Cinner et al., 2008; Davies et al., 2009; Welcomme et al., 2010). Fishing occurred at many locations across each type of habitat and was generally widely dispersed. Fishing occurred regularly within the strict conservation zone in the marsh and no-take zones at the edge of the lake. Compliance with gear restrictions was also generally low and fishers used mesh sizes below the legal minimum primarily out of perceived necessity. Although this is common in heavily exploited artisanal fisheries, using progressively smaller mesh sizes leads to smaller stocks and smaller catches (Eggert and Lokina, 2010). Accordingly, in absence of data for lengths of fish caught, consistent declines in mesh sizes can also be used as an indicator of the state of the fishery (Allan et al., 2005). To inform management, further research is needed to determine factors affecting catch size and fisher spatial behaviour at Lake Alaotra. Implementing conservation and no-take zones without adequately consulting fishers or understanding their spatial distribution will compromise the potential effectiveness of management actions (Daw, 2008).

Although a wide range of species are currently observed in fishers' catches, the series of fish introductions undertaken at Lake Alaotra and fishing pressure has irreversibly changed the fishery and assemblage of species present. In contrast to catch compositions observed by Moreau in 1976 (see Moreau, 1979a), current catches are dominated by introduced and highly invasive Nile tilapia and blotched snakehead. Nile tilapia tolerate a wide range of often poor environmental conditions and typically outcompete native species (Canonico et al., 2005); interestingly, Moreau (1979a) predicted the dominance of Nile tilapia in Lake Alaotra. Although carp was also introduced (to create a food fish fishery), its decline in catches since 1976 has probably compromised the income and livelihood of fishers considerably; carp is highly valued (Copsey et al., 2009), particularly during the wet season when other high-value fish such as large Nile tilapia are in shorter supply. It is common for the composition and commercial focus of a fishery to change with fish introductions; this has also been observed in other freshwater systems in Africa (Pitcher, 1994), including Lake Victoria (Balirwa et al., 2003). While these changes do not occur overnight, they can have profound effects on native fish species and biodiversity, as well as local people reliant on the fishery for livelihood (Canonico et al., 2005).

The Lake Alaotra fishery appears to be moving toward a two-species system; few species other than Nile tilapia and blotched snakehead are observed in large quantities. The latter is implicated in the decline or disappearance of native birds, such as the Madagascar pochard (*Aythya innotata*) and Alaotran grebe (*Tachybaptus rufolavatus*), as well as native fish species and the black bass (still found in some areas of the lake) (Wilmé, 1994; Pidgeon, 1996; Hawkins et al., 2000; Benstead et al., 2003). The crayfish (*Procambarus alleni*) recently observed in Lake Alaotra is also a threat to freshwater biodiversity and local livelihoods. Although its potential impact on the Lake Alaotra ecosystem can only be estimated, this exotic species is highly fecund, could potentially outcompete native *Astacoides* crayfish as well as freshwater fish species, and may damage rice crops (Jones et al., 2009). Although neither the crayfish nor blotched snakehead can be eradicated, management strategies are required to control their proliferation (Courtenay et al., 2004; Jones et al., 2009).

Differences in catch composition across habitats and seasons may be related to differences in species' abundance or catchability due to seasonal variation (Forcada et al., 2009). Lower water level during the dry season provides opportunities to use some methods (such as rod & bubble) more frequently than at other times of the year; it also enables people to enter the fishery as casual (itinerant) fishers. Fishers also indicated that fish were easier to catch when water level is low, which explains the apparent increase in fishing intensity observed late in the dry season. This study indicates that differences in catch composition can stem from shifts in fisher behaviour due to seasonal factors (such as water level) and suggests that catch is a function of species ecology and fisher distribution.

Average size of fish caught also varied between seasons and across years. Fish caught were generally longer and heavier during the dry season than the wet season. Seasonal differences in fish sizes are probably primarily due to species-specific reproductive cycles and growth patterns (see Moreau and Moreau, 1987; Froese and Pauly, 2010). The inter-annual differences in fish size probably stem from the highly stochastic nature of the floodplain. Fishery productivity, including better fish reproduction and growth, tends to be greater in wetter years (Halls et al., 2011; Kolding and van Zwieten, 2012). Conversely, dry years have the dual impact of poor recruitment and growth as well as greater fishing mortality because fish are more easily caught (Halls and Welcomme, 2004). Because the prices fishers receive for their catches are determined mainly by fish size, it is likely that catch values and fishers' income will fluctuate over the year as well as between years. This will reduce predictability of fishers' income, which could impact their livelihood. The generally smaller lengths and weights of fish during the wet season may make this a period of relative hardship for fishers.

The timing of fish reproduction observed during this study highlights the mismatch between current management policies and fish biology. Although the current timing of the closed period (15 November to 15 January) partly corresponds with increased availability of work in rice fields, it provides little or no protection for spawning fish. An obvious solution is to adjust the timing of the closed period to coincide with fish reproduction; however, potential impacts on fishers must be considered. If the closed period is brought forward it should also be linked with a local work program to reduce the impacts on fisher livelihood; for example, hiring fishers to (i) remove invasive plants, (ii) demarcate mutually agreed no-take zones, or (iii) plant trees on hills to the east of the lake to reduce erosion. The levy on fish sold to collectors currently collected by the Federation of Fishers could be used to fund such a program. This will help to ensure fishers have an alternative source of income during the closed period and increase the likelihood of compliance.

Despite relatively small catch sizes, the Anororo fishery is highly commercial and cash driven. Catch sizes differed significantly over months of the study, suggesting that income from fishing will also vary throughout the year. Although some species have greater cash value than others (Copsey et al., 2009b), the methods used most by fishers (traps and gill nets) generally do not target specific species. Fishers often kept rather than sold large Nile tilapia or carp (the two most valuable species) on the relatively rare occasions when they were caught, indicating that Anororo-based fishers do not always maximise profits. This behaviour is contrary to the expectations of traditional micro-economic theory (Stigler and Becker, 1977; Thaler, 2000). With half of catches less than 16 fish or 2kg in weight, Anororo-based fishers may earn only US$1.36 per day, only marginally above the internationally-accepted extreme poverty standard of US$1.25 a day (World Bank, 2008). Although stabilising commodity prices is often a policy option (Minten et al., 2006), increasing or stabilising fish prices would probably increase fisher effort and hence considerably deplete fish stocks (Munro, 1983).

Increased fishing effort in multi-species fisheries such as Lake Alaotra typically leads to a measureable decrease in average size for each species caught (Welcomme, 2001). Records of the lengths of fish caught can therefore be used to indicate fishing pressure over time and gauge fishery status (Welcomme, 2001). Quantitative and qualitative evidence shows that the mean size of fish caught in Lake Alaotra has decreased progressively. Length-frequency distributions for Nile tilapia and blotched snakehead confirm overfishing (Froese, 2004), which is one of the most significant threats to freshwater systems in Madagascar (Benstead et al., 2003). The high proportion of Nile tilapia caught under legal size is also an indicator of low levels of compliance with fishing regulations. Additionally, none of the fish caught during the study had attained maximum species-specific length and few could be classed as large. In a healthy fishery catches can be expected to contain at least 20% large fish (Froese, 2004), which, as megaspawners, usually produce disproportionately more eggs and fry than smaller individuals (Blueweiss et al., 1978). These findings suggest that a maximum size limit to protect highly fecund individuals might benefit Lake Alaotra's fishery, particularly for commercially-important species such as Nile tilapia. Interviews with fishers suggest that fish caught in Lake Alaotra are smaller each year, because larger fish are fished out and juvenile fish do not have sufficient time to grow before being caught. This reduction in fish sizes is one of the primary indicators of overfishing (Welcomme et al., 2010).

Declining annual fish production, catch size, and fish size over time are characteristic patterns of a fishery in decline (Allan et al., 2005). Given the large number of fishers and high fishing effort at Lake Alaotra, as well as anecdotal evidence of declining catches and fish sizes, the increasing human population, open-access nature of the fishery, and limited land available for further rice fields are likely to add extensive pressure on marsh and lake resources, and accelerate the fishery's decline (see Durand, 1979; Amarasinghe and de Silva, 1999; Balirwa et al., 2003; Béné et al., 2003; Matsuishi et al., 2006; Eggert and Lokina, 2010). Overfishing in Lake Alaotra threatens one of the area's core livelihoods and therefore the wellbeing of many households. Present levels of exploitation from the lake and marsh are not sustainable and may lead to collapse of the fishery within the next decade if appropriate management action is not undertaken.

Key findings from this study that can improve management of the fishery are that (i) many fishers are highly dependent on the fishery for livelihood throughout the year despite unpredictability and variability in catch, (ii) fisher behaviour varies over the year in an attempt to maintain catch, including using a range of fishing methods and locations, (iii) catch composition varies across habitats and between seasons, (iv) catches and therefore income from fishing is highly variable between and within fishers on daily and seasonal bases, (v) a high proportion of fish are caught before reaching maturity, and (vi) fishers perceive the fishery to be in a state of decline. The results also suggest that water level may be a key factor in selecting fishing locations and/or methods used, and could provide insights about the spatial distribution of fishing activity throughout the year.

This comprehensive understanding of the fishery provides a foundation for further analyses assessing the drivers of catch size and fisher spatial behaviour, as well as the short-term costs to fishers of alternative management plans. Incorporating spatially explicit knowledge of fishing behaviour is particularly important to promote stakeholder receptiveness and compliance when introducing a management structure and establishing regulations (Daw, 2008). The highly variable nature of the fishery indicates that management should be adaptive; use of novel approaches, such as flexible spatial and temporal closures suggested by Game et al. (2009) and Grantham et al. (2008) for marine fisheries, may also be required. Effective biodiversity conservation and management of the Lake Alaotra wetland will require coordinated planning that considers fishers and other stakeholders who rely on and influence use of its natural resources.

CHAPTER 5 PREDICTING THE POTENTIAL IMPACTS OF CONSERVATION INTERVENTIONS ON FISHERS

This chapter has been submitted for publication as: Wallace, A.P.C., E.J. Milner-Gulland, J.P.G. Jones, N. Bunnefeld, R.P. Young, and E. Nicholson. Predicting the potential impacts of conservation interventions on fishers. *Biological Conservation*. Andrea Wallace developed the research questions, collected data, conducted analysis and developed the models, prepared drafts and the final paper, led the editing process, and maintained editorial control over co-author contributions.

5.1 INTRODUCTION

Natural resource management plans often fail to address the human dimensions of conservation (Margules and Pressey, 2000; Heemskerk et al., 2003; Berkes, 2004). Short-term costs to resource users of conservation interventions are rarely quantified, and their heterogeneities even less so, yet understanding and mitigating these costs is crucial for effective conservation planning (Margules and Sarkar, 2007; Levin et al., 2009; Mascia et al., 2010). Attaining more realistic and accurate estimates of cost for resource users in order to improve resource management plans can help to minimise impacts on resource users and lead to better compliance (Carwardine et al., 2010; Eggert and Lokina, 2010). Many managers now recognise that understanding stakeholder behaviour may be the most important factor for long term fishery sustainability (see Bunnefeld et al., 2011; Fulton et al., 2011). Indeed, misunderstanding resource use and stakeholder behaviour may be one of the main causes of many management failures (Berkes, 2003; Mills et al., 2010; Weeks et al., 2010b).

Artisanal fisheries in developing countries are diverse and dynamic (Abernethy et al., 2007; Maccord et al., 2007; McClanahan et al., 2008b). Spatial and temporal variations in resource availability, and in the types of gear used, are likely to lead to variation in catch size. This diversity means that management interventions, such as conservation or no-take zones, seasonal closures, and the minimum length of fish that may be caught, are likely to have differential impacts on returns from fishing for different groups of fishers (Welcomme et al., 2010). Accordingly, to develop appropriate management policies it is critical to know the livelihood impacts of conservation interventions for different groups of resource users (Allison and Ellis, 2001; Smith et al., 2005; Welcomme, 2011a). Most inland fisheries are multi-species, multi-gear fisheries where traditional methods of stock evaluation are inappropriate (Welcomme, 2001). Catch per unit effort (CPUE) data based on experimental fishing records collected over time can be used as an index of relative fish stock abundance. However, for artisanal fisheries such data are often not available and, in these cases, catch and effort data from fisher reports can be used to understand trends in resource value to users (Welcomme, 2001).

The Lake Alaotra wetland is the primary inland fishery in Madagascar and a site of biodiversity conservation importance, providing the only habitat for the Critically Endangered Alaotran gentle lemur (*Hapalemur alaotrensis*). Local median income from fishing is approximately US$1.36 per day (see Chapter 4), which only slightly exceeds the standard measure for extreme poverty of US$1.25 per day (World Bank, 2008). The floodplains surrounding the wetland are also a key rice production area and the marsh continues to be converted to rice agriculture and degraded through burning and harvesting of reeds (Mutschler et al., 2001; Andrianandrasana et al., 2005; Ralainasolo et al., 2006; Copsey et al., 2009a). New conservation interventions for Lake Alaotra are currently in the advanced planning stage, making it a perfect case study to assess diversity in gear, effort, and catch size in artisanal fisheries, and consequent variation in the potential impacts of any restrictions to fishing activity. The planned interventions are based on the 2006 Lake Alaotra Management Plan, developed in collaboration with the regional Fisheries Service by the Durrell Wildlife Conservation Trust (DWCT). These interventions include a strict conservation zone in the centre of the marsh to protect habitat for lemurs and threatened wetland birds, as well as no-take zones around the lake edge to protect fish spawning aggregations; together these are referred to as 'restricted areas' (Figure 5.1; Razanadrakoto and Rafaliarison, 2005). These spatial interventions are additional to existing regulations, including gear restrictions, minimum fish size limits, and an annual two-month fishery closure (Dina de Pêche, 2006), which collectively aim to achieve a sustainably managed fishery and reduce pressures on wetland species of conservation concern.

Attempts to implement and enforce the management plan and fishing regulations for Lake Alaotra have not been successful thus far. The regional Fisheries Service has twice demarcated no-take zones at the lake edge and each time the markers have been removed. It has also been difficult to control and enforce gear restrictions in a fishery where fishers construct their own equipment (B-J. Rasolonjatovo, DWCT, pers. comm.). Fishers use locations across a range of habitats, including open lake, lake edges, and marsh, and many of these locations are within the planned restricted areas (Figure 5.1). Furthermore, current timing of the closed period (15 November to 15 January) is mis-matched with the timing of fish recruitment. Fish eggs and fry were observed by AW in catches from late September to mid-November, peaking during October. Approximately 33% of interviewed fishers also indicated that the closed period should occur earlier (e.g., from 1 October to 30 November) to protect fish laying eggs as well as juvenile dispersal (see Chapter 7).

At the time the management plan was developed, little information and no systematic data regarding fishers' spatial and temporal behaviour existed, and hence the costs of interventions to fishers were poorly accounted for. These costs may be very high due to the large number of fishers and their heavy dependence on the lake for their livelihoods (Wilmé, 1994; Pidgeon, 1996), potentially making compliance difficult and eventually leading to failure of the management plan.

The aim of this study was to assess how two management interventions, restricted areas and an earlier temporal closure (October-November), map onto patterns of current fishing activity and to explore potential impacts on fishers of compliance with these interventions. In particular, I examined how spatial and temporal patterns of catch size and effort vary across the two primary methods used in the fishery, traps and gill nets, in order to evaluate the potential differential impacts of the two management interventions on the behaviour and livelihoods of fishers using these methods.

5.2 METHODS

5.2.1 Study site

Please refer to Section 3.1 in Chapter 3 for details of the study site. Fishers at this site use a variety of habitats, fishing methods, and fishing locations (Figure 5.1).

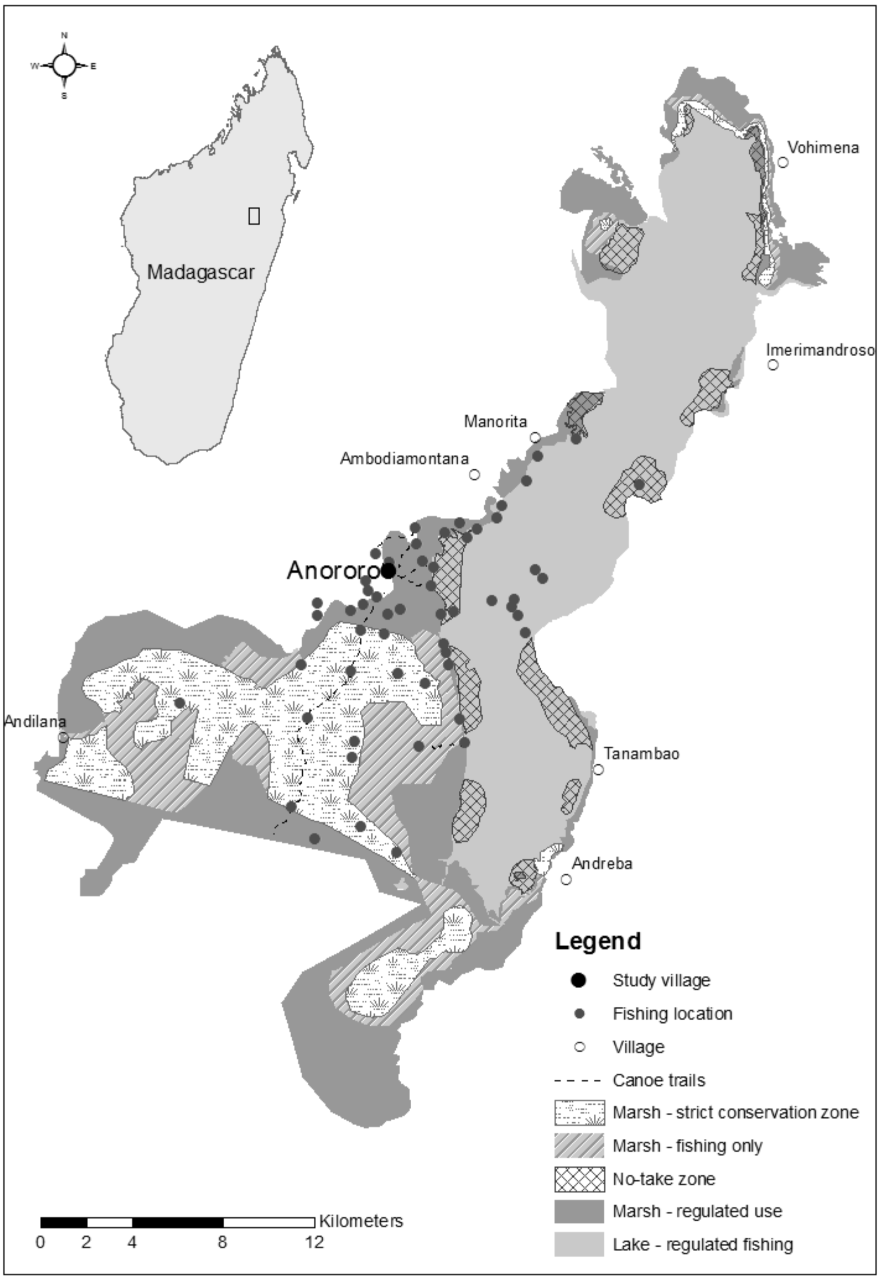


Figure 5.1. Map of Lake Alaotra showing planned management zones within the lake and adjacent marsh, and the centroids of fishing locations used by local fishers as recorded in the catch monitoring data. Planned restricted areas where fishing is to be prohibited are the strict conservation zone in the centre of the marsh and no-take zones around the lake edge.

5.2.2 Data collection

Please refer to Section 3.2 in Chapter 3 for an overview of general data collection methods. Methods specific to this chapter are provided below.

Data were collected in June and July 2009 and for 14 months from October 2009 to December 2010, comprising two dry seasons (2009 and 2010) and one wet season (2010). Structured catch interviews (n = 1,800), including measurement of fish lengths (n = 27,064), were conducted with Anororo-based fishers returning from their fishing trip. Catch interview data were not collected during December in either year because this coincided with the current annual fishery closure. A total of 537 individual fishers participated in catch interviews; 248 (46%) were interviewed more than once. I assigned respondent codes to fishers to preserve their anonymity (Bernard, 2002). Catch interviews were 5 to 10 minutes in duration, comprising (a) questions to determine the fisher's fishing location, gear used, and effort that day and (b) counting and measuring fish caught. Species-specific length-weight relationships, calculated from a sample of 498 fish lengths and weights recorded during the study, were used to estimate weights of fish measured during catch interviews as well as total weight for each fisher's catch.

5.2.3 Data analysis

5.2.3.1 Defining catch and fisher effort

Although the equation for determining catch per unit effort (CPUE = Catch/Effort) is not complex, defining and estimating both catch and effort within multi-species and multi-gear fisheries can be challenging. Fish catches in Lake Alaotra can be highly variable on a daily and seasonal basis. Because catches typically consist of a mix of species, the value of the catch primarily depends on fish size rather than species. Larger fish weigh more and attract a higher price. I assessed catch as total weight for the fishing trip, reflecting economic value for the fisher and the common perception that fishers are profit maximisers (Tidd et al., 2011).

Although Anororo-based fishers use a variety of fishing methods, traps and gill nets are predominant and were reported during 91% of catch interviews. Fidelity to each gear type is high; only 2.6% of fishers used both traps and gill nets during the study period, but without temporal overlap. I analysed each of these two gear types separately and used economically relevant variables (i.e., travel time to fishing location, time spent fishing, and number of gear items used) rather than biological variables (i.e., soak time – the time fishing gear is in the water) to define fisher effort.

Rist et al. (2008) show that estimating effort of bushmeat hunters is problematic because multiple methods are used to target different species, equipment is checked at variable rates, and total time measures can be biased and overestimate biologically relevant effort. In my case, using the fishing trip as the unit of effort was more meaningful when assessing management options from a fisher's perspective. It provided a realistic representation of catch value and effort invested in fishing, allowing examination of the potential impacts of conservation interventions on fishers. Although the total length of a trip varied between fishers, total trip time is included in other effort variables in the model. Travel time to a fishing location is significantly and strongly positively correlated with total trip time for trap fishers (r = 0.81, $p < 0.01$), while time spent fishing is equivalently correlated with total trip time for gill net fishers (r = 0.85, $p < 0.01$). These measures of effort were used to account for variation in total trip time.

5.2.3.2 Data preparation

I analysed data for 1,284 fishing trips by 284 trap fishers and 319 trips by 158 gill net fishers. In 33 cases, no fish were measured because fishers had already sold their catch prior to interview, did not want to have their fish measured, or did not have time for their fish to be measured; these interviews were excluded from analyses. Interviews with fishers who had fished within village border habitat were infrequent ($n_{traps} = 3$; $n_{gill\ nets} = 1$) and also excluded from analyses. Not all combinations of year and month were covered in the data and so I was not able to include year and month as separate categorical variables. To account for potential effects of different years and months on catch size, months were grouped primarily in terms of water level, but also considering rainfall and season, as well as rice cultivation activities that affect level of effort put into fishing (Table 5.1). The grouped months were then combined with year to achieve a single 'Time Period' categorical variable with eight levels.

Table 5.1. Variables used to categorise months into groups. Months in bold are those where data was collected in both years (2009 and 2010). Water level categories reflect the average range of values above the mean lowest level: Highest = +1.6m to +2.0m; High = +0.9m to +1.69m; Medium = +0.2m to +0.89m; Low = 0m to +0.19m. Mean rainfall: High >150mm; Medium = 30mm to 149mm; Low = 10mm to 29mm; Very low <10mm.

Group	Month	Water level[a]	Rainfall[b]	Season	Rice cultivation activities[c]
Jan-Feb	January	High – increasing	High	Wet	1st season planting
	February	High – increasing	High	Wet	1st season planting
Mar-Apr	March	Highest	Medium	Wet	1st season maintenance
	April	Highest	Medium	Wet	1st season maintenance
May-Jun	May	High – decreasing	Low	Dry	1st season harvesting
	June	High – decreasing	Very low	Dry	1st season harvesting
Jul-Sep	**July**	Medium – decreasing	Very low	Dry	2nd season planting
	August	Medium – decreasing	Very low	Dry	2nd season planting
	September	Medium – decreasing	Very low	Dry	2nd season planting
Oct-Nov	**October**	Low – still decreasing	Medium	Dry	2nd season maintenance; 1st season prep.
	November	Lowest	Medium	Dry	2nd season maintenance; 1st season prep.
n/a	December[d]	Medium – increasing	High	Wet	2nd season harvest; 1st season prep.

[a] Source: this study and Ferry (2009).
[b] Source: Unpublished data from BVLac, Madagascar. Average rainfall between 1964 and 2009.
[c] Source: this study and Le Courtois (2010).
[d] Catch interviews were not conducted during the month of December. If data were available, and based on the above criteria, December would be grouped with Jan-Feb.
n/a = not applicable.

5.2.3.3 Analysis of catch size

I used linear mixed effects models (LMMs) to identify factors influencing catch size for fishers using traps or gill nets. This approach accommodates a wider range of data with different types of response and explanatory variables, as well as random effects, which are all common attributes of ecological datasets (Faraway, 2006; Zuur et al., 2009). One type of random effect is variation between individual fishers, when multiple responses are measured per individual, such as recurring trips by individual fishers or fishing vessels (Venables and Dichmont, 2004b). Despite the fact that fisheries data are often structured in this way, the use of mixed models in fisheries research is relatively recent (Helser et al., 2004; Venables and Dichmont, 2004a, b).

The response variable for both the trap and gill net models was catch size – total weight of catch from the fishing trip. To account for differences over time in fisher effort, gear characteristics, and spatial variability, I included time of year, travel time to fishing location, time spent fishing, number of gear items used, their size and mesh size, and the habitat and restricted status of the fishing location as explanatory variables (fixed effects) in each model (Table 5.2). Although the models are not geographically explicit, by including travel time to fishing location, whether the fishing location is inside or outside of an area with restricted status in the management plan, and habitat as fixed effects, the models are spatially implicit. To account for differences among fishers, individual fishers were included as a random effect in both models (Table 5.2).

The models included a mixture of categorical and continuous variables. Continuous variables were normalised using log transformations. Because catch size was zero in a very few cases (n_{traps} = 10; $n_{gill\ nets}$ = 3), I added a constant of one gram to each catch size before log-transformation. Interactions between explanatory variables were explored visually and none were found; there was also no *a priori* reason to expect interactions. Log-transformed values for catch size were also used for analysis of the impacts of spatial and temporal restrictions.

Pearson's correlations were used to test for collinearity among fisher effort variables and gear characteristics. Gear and mesh size were either not significantly correlated (*r*

< 0.5, $p > 0.05$) or only weakly correlated ($r < 0.3$, $p < 0.05$) with fisher effort variables in each dataset. Although number of traps used and time spent fishing were moderately correlated overall ($r = 0.57$, $p < 0.05$), if tested at individual fisher level, some were strongly positively correlated ($r = 0.8$) while others were minimally ($r = 0.1$) or negatively ($r = -0.3$) correlated. This variation and absence of a clear relationship between number of traps used and time spent fishing justified retaining both variables as fixed effects.

Number of gill nets used and time spent fishing were weakly negatively correlated ($r = -0.17$, $p < 0.05$), which can be attributed to variation among fishers as well as the manner in which gill nets are used. Gill nets can be used in three ways: passively overnight, passively while waiting, or actively. Differences in manner of use are encompassed by other effort variables including number of gear items used and time spent fishing. For example, fishers using gill nets actively typically use fewer nets but may fish for highly variable, and often extended, periods whereas fishers using gill nets passively overnight may have many nets but require little time to check them for fish.

Table 5.2. List, type, and description of variables used to predict catch size in two separate LMMs for trap and gill net fishers.

Variable	Type	Description
Response variable		
Catch	Continuous	Grams of fish caught during a fishing trip at a given location (location is as defined by fishers).
Explanatory variables (fixed effects)		
TimePeriod	Categorical	8-level factor; combines year (2009 or 2010) and months grouped according to water level, rainfall, season, and timing of rice cultivation activities.
TimeTo	Continuous	Estimated travel time in minutes from the village to the fishing location.
TimeFishing	Continuous	Estimated time in minutes spent fishing (removing fish from traps or gill nets after arriving at the fishing location and before beginning to travel back to the village).
NumberUsed	Continuous	Estimated number of gear items used on the trip.
Size	Continuous	Size of gear item: average volume of a trap (m^3) or average area of a gill net (m^2).
Mesh	Continuous	Mesh size in millimetres.
Habitat	Categorical	2-level factor (traps: marsh and lake-marsh edge) or 3-level factor (gill nets: marsh, lake-marsh edge, and lake); traps are not used in lake habitat. Fishing location used on a trip is within a single habitat.
Restricted	Categorical	2-level factor; protected area status: fishing location is inside (1) or outside (0) of the core protected area in the marsh or the no-take zones (NTZs) at the lake edge.
Random effect		
FisherID	Categorical	Individual fishers were identified by a unique ID and may have multiple fishing trips within the datasets.

5.2.3.4 *Statistical inference*

Information-theoretic approaches, such as AIC (Akaike's Information Criterion; Akaike, 1974), use measures of predictive power to rank models and quantify the magnitude of difference between models (Burnham and Anderson, 2002; Bolker et al., 2009). Rather than selecting the 'best' model, information-theoretic tools are used to select and average candidate or well-fitting models (Bolker et al., 2009). This approach is also robust to the mild to moderate degree of collinearity found in this study (Freckleton, 2011).

I used AIC model selection and model averaging to determine the relative importance and averaged estimates for each variable. Analyses were performed in R version 2.13.1 (R Development Core Team, 2011). Global models (fitted by maximum likelihood) were run using the lme4 package in R; the MuMIn package was used for model comparison and model averaging. Following Burnham and Anderson's (2002) rule of thumb, all models where AIC differences were <4 (traps) or <6 (gill nets) were included in the candidate set of models for model averaging. AIC differences of <4 and <6 were chosen because the weight or support for subsequent models decreased considerably at this point in the trap and gill net model selection tables, respectively. No single model was clearly superior to others in the candidate set of models for either gear type, suggesting that model averaging would provide a more robust understanding of the system and reduce model selection bias effects (Burnham and Anderson, 2002).

5.3 RESULTS

Catch size varied between habitats and gear types, between areas with different planned restricted status, and on a temporal basis, as well as between individuals (Figures 5.2 and 5.3). Fishers using traps fished almost exclusively in marsh or edge habitat, while gill net fishers primarily fished in open water (Figure 5.2). Plots of catch sizes in relation to fisher effort, measured as number of gear items used, time spent fishing, and travel time to fishing location, indicated that catch size generally increases with increasing fisher effort (Figure S5.1 in Appendix S5).

5.3.1 Factors affecting catch size

The most parsimonious models for trap fishers are shown in Table 5.3. The model with the lowest AIC and most support (W_i = 0.14) indicated that the combined effects of the restricted area status of the fishing location, time of year, and various measures of fisher effort influenced catch size for trap fishers. Model averaging indicated that the four most important variables were restricted area status, time of year, number of gear items used, and time spent fishing. These four variables had the highest variable weight (w_i = 1.0; Table 5.3) and the strongest effect on catch size (see Figure 5.4a). Gear size and travel time to fishing location had comparatively lower variable weights (0.61 and 0.53, respectively) across all models.

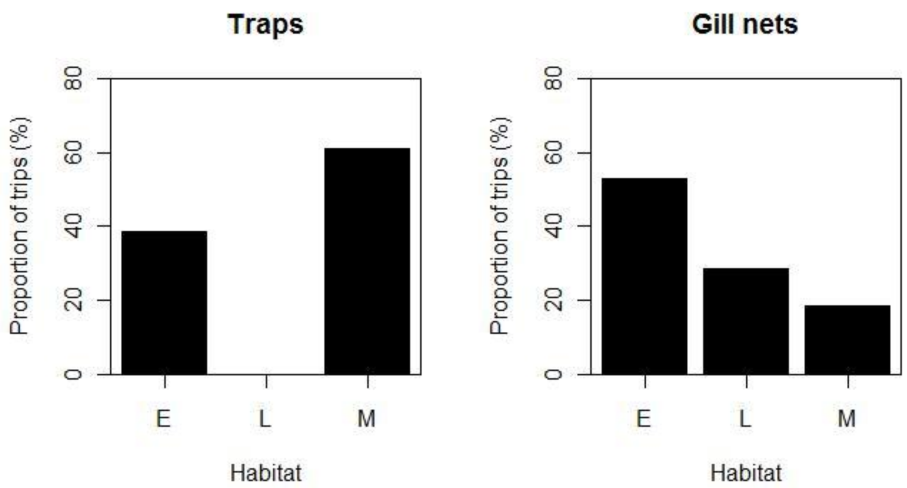


Figure 5.2. Proportion of fishing trips by habitat for each gear type; E = Lake-Marsh Edge, L = Lake, M = Marsh.

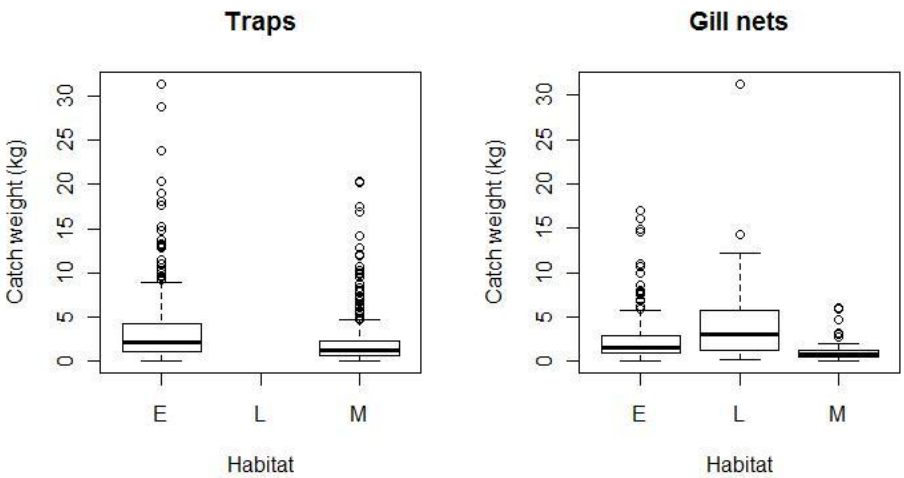


Figure 5.3. Box and whisker plots of raw catch weight (kg) by habitat for each gear type; E = Lake-marsh Edge, L = Lake, M = Marsh. The horizontal bar represents the 50th percentile, the top of the box the 75th percentile, and the base of the box the 25th percentile. Whiskers represent the range of data, and open circles are outliers.

The gill nets model with the lowest AIC and most support (W_i = 0.26) indicated that the combined effects of time of year and fisher effort influenced catch size for gill net fishers (Table 5.4). Model averaging indicated that the four most important variables were time of year, number of gear items used, travel time to fishing location, and time spent fishing. These four variables had the highest variable weight (w_i = 1.0; Table 5.4) and the strongest effect on catch size (see Figure 5.4b); gear size had a variable weight of 0.70. In contrast to trap fishers, the restricted area status of fishing locations was not an important factor influencing catch size for gill net fishers. Catch sizes in relation to restricted area status for each of the models with lowest AIC show differences between gear types (Figure S5.2). Effort factors recurring in both models were time spent fishing and number of gear items used. The random effect of individual fishers represented 27% and 24% of the variance in the traps and gill nets models with lowest AIC, respectively.

There was also inter-annual and intra-annual variability in catch size per trip, for both trap fishers and gill net fishers (Figure 5.5). Gill net fishers experienced greater temporal variation in catch size than trap fishers. Catch sizes for trap fishers were largest in the Jan-Feb month group (wet season) and smallest in the May-Jun and Jul-

Sep groups (dry season), representing a 45% decrease in mean catch size from best to worst time periods (Figure 5.5a). In contrast, catch sizes for gill net fishers were largest in the Oct-Nov group (dry season), and smallest in Jan-Feb (wet season), representing a 74% decrease in mean catch size between the best and worst time periods (Figure 5.5b). The high catches in Oct-Nov were a function of a high catch coefficient (Table S5.1 in Appendix S5) and increased time spent fishing in those months. Focus group sessions and interviews with fishers confirmed fish are easier to catch with gill nets during the dry season because water levels are lower. Accordingly, gill net fishers spend more time fishing at that time of year to maximise catch size and income.

Table 5.3. Coefficients for the fixed effects of the 16 most parsimonious models that were used in model averaging for the traps model. A '+' indicates that a factor variable was included in the model, whereas a blank field means that the variable was not included. Coefficients cannot be presented for factor variables (see Table S5.1 in Appendix S5 for averaged model parameters). The number of parameters in the model (k), the AIC and AIC difference (ΔAIC), and weight (W_i) is given for each model. Individual variable weights (w_i) are also provided.

Intercept	Restricted	TimePeriod	Number Used	Time Fishing	TimeTo	Size	Habitat	Mesh	k	AIC	ΔAIC	Model weights (W_i)
3.096	+	+	0.6991	0.2664	0.0700	0.1136			15	3595	0.00	0.14
3.234	+	+	0.7183	0.2736		0.1158			14	3595	0.28	0.12
3.274	+	+	0.6806	0.2626	0.0684	0.1113	+		16	3596	0.88	0.09
2.820	+	+	0.7131	0.2654	0.0716				14	3596	0.90	0.09
3.416	+	+	0.6986	0.2695		0.1132	+		15	3596	1.05	0.08
2.956	+	+	0.7328	0.2728					13	3596	1.29	0.07
3.012	+	+	0.6935	0.2614	0.0698		+		15	3597	1.66	0.06
3.446	+	+	0.6947	0.2678	0.0701	0.1135		-0.1037	16	3597	1.83	0.06
3.153	+	+	0.7119	0.2684			+		14	3597	1.93	0.05
3.579	+	+	0.7140	0.2750		0.1157		-0.1023	15	3597	2.12	0.05
3.173	+	+	0.7087	0.2667	0.0717			-0.1048	15	3598	2.73	0.04
3.535	+	+	0.6778	0.2638	0.0685	0.1113	+	-0.0791	17	3598	2.80	0.03
3.668	+	+	0.6960	0.2707		0.1133	+	-0.0765	16	3598	2.98	0.03
3.304	+	+	0.7285	0.2742				-0.1032	14	3598	3.12	0.03
3.272	+	+	0.6908	0.2625	0.0699		+	-0.0788	16	3598	3.59	0.02
3.404	+	+	0.7093	0.2696			+	-0.0761	15	3599	3.86	0.02
Individual variable weights (w_i)	1.00	1.00	1.00	1.00	0.53	0.61	0.40	0.28				

Table 5.4. Coefficients for the fixed effects of the 14 most parsimonious models that were used in model averaging for the gill nets model. A '+' indicates that a factor variable was included in the model, whereas a blank field means that the variable was not included. Coefficients cannot be presented for factor variables (see Table S5.1 in Appendix S5 for averaged model parameters). The number of parameters in the model (k), the AIC and AIC difference (ΔAIC), and weight (W_i) is given for each model. Individual variable weights (w_i) are also provided.

Intercept	Restricted	TimePeriod	Number Used	Time Fishing	TimeTo	Size	Habitat	Mesh	k	AIC	ΔAIC	Model weights (W_i)
2.835		+	0.2469	0.2453	0.3787	0.2557			14	945.9	0.00	0.26
2.858	+	+	0.2471	0.2417	0.3810	0.2527			15	947.6	1.95	0.10
2.353		+	0.2395	0.2450	0.3700	0.2403		0.1643	15	947.7	2.05	0.09
3.272		+	0.2379	0.2576	0.3066	0.2328	+		16	947.6	2.15	0.09
3.569		+	0.2936	0.2795	0.4180				13	948.3	2.17	0.09
2.301		+	0.2698	0.2741	0.3911			0.3954	14	949.3	3.36	0.05
3.997		+	0.2789	0.2899	0.3300		+		15	949.2	3.47	0.05
2.322	+	+	0.2390	0.2411	0.3714	0.2353		0.1834	16	949.5	3.97	0.04
3.350	+	+	0.2379	0.2570	0.3043	0.2301	+		17	949.2	3.99	0.04
3.585	+	+	0.2937	0.2754	0.4198				14	949.9	4.04	0.03
2.927		+	0.2329	0.2574	0.3013	0.2222	+	0.1158	17	949.6	4.31	0.03
2.941		+	0.2605	0.2855	0.3118		+	0.3241	16	950.5	5.03	0.02
2.262	+	+	0.2692	0.2693	0.3917			0.4130	15	950.8	5.14	0.02
4.074	+	+	0.2783	0.2891	0.3271		+		16	950.7	5.23	0.02
Individual variable weights (w_i)	0.26	1.00	1.00	1.00	1.00	0.70	0.26	0.27				

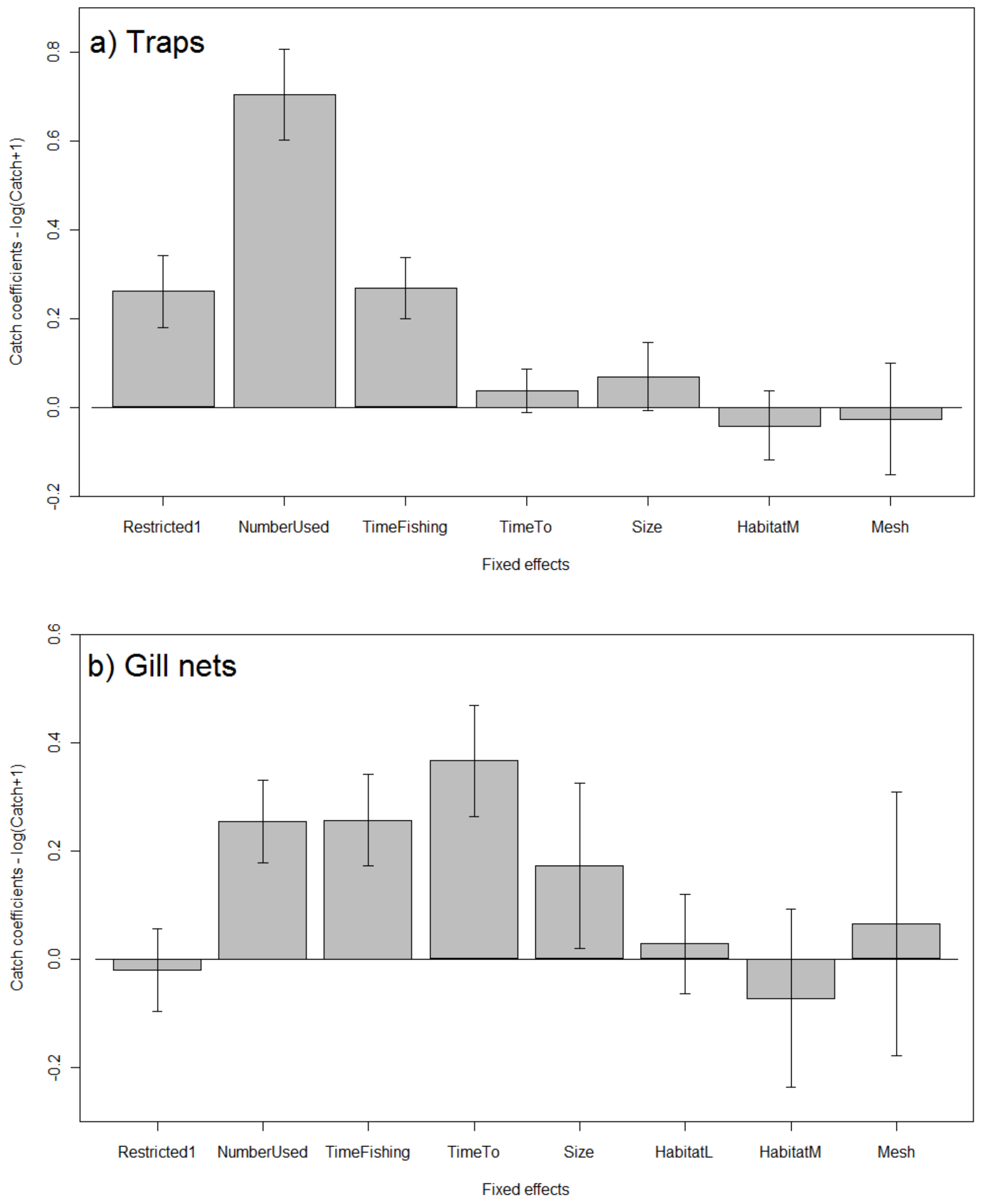
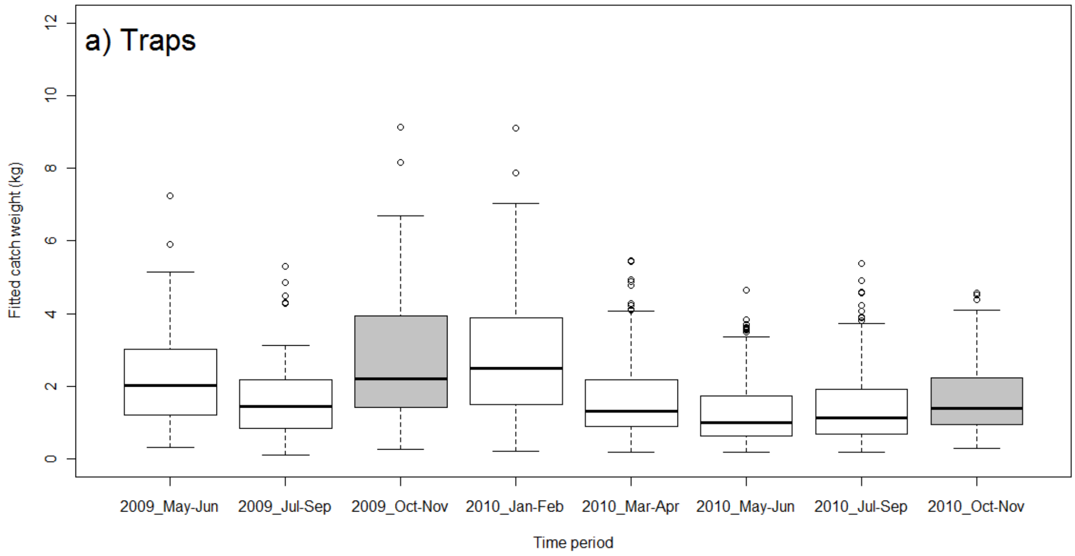


Figure 5.4. Coefficient averages (± 1SE) from the candidate set of models for a) traps and b) gill nets, explaining the variation in catch size as influenced by restricted area status (restricted versus non-restricted), fisher effort, habitat, and gear characteristics. Restricted1 = restricted area; NumberUsed = number of gear items used; TimeFishing = time spent fishing; TimeTo = travel time to fishing location; Size = gear size; HabitatL = lake habitat; HabitatM = marsh habitat; Mesh = mesh size. Baseline (i.e., zero line) levels for restricted and habitat variables are 'non-restricted' and 'edge', respectively. See Table 5.2 for variable descriptions and Table S5.1 in Appendix S5 for coefficient values.



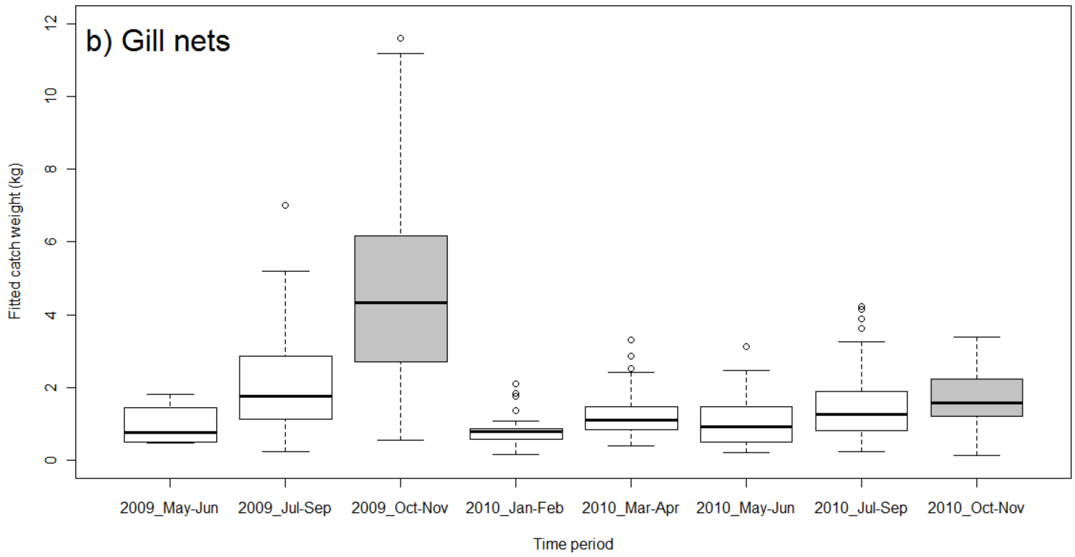


Figure 5.5. Fitted catch weight for the top model based on lowest AIC for each gear type. Box and whisker plot of modelled catch weight per trip in kilograms for a) trap fishers and b) gill net fishers over the study period; the horizontal bar represents the 50th percentile, the top of the box the 75th percentile, and the base of the box the 25th percentile. Whiskers represent the range of data, and open circles are outliers. Catch sizes are largest in Jan-Feb for trap fishers and in Oct-Nov for gill net fishers. Time periods with smallest catch size are May-Jun and Jul-Sep for trap fishers and Jan-Feb for gill net fishers. Grey shading refers to the proposed earlier closed period; the current closed period is not represented in the data.

5.3.2 Impacts of spatial and temporal restrictions by gear type

Trap fishers fishing in proposed restricted areas had a larger mean catch size than those in non-restricted areas (Table S5.2; also see Figure S5.2). Mean catch size was largest during October-November compared to the rest of the year as well as the study period as a whole for both gear types, particularly for gill net fishers (Table S5.2; also see Figure 5.5). I used this information to estimate the maximum costs that could be incurred by fishers due to either or both of the restrictions, based on two simplifying conditions: first, I did not account for potential adaptive changes in fisher behaviour in response to the interventions; and second, I did not account for potential changes in fish biomass resulting from the interventions. I focused on immediate short-term impacts for fishers. Fishers are likely to adapt to reduce impacts, although with varied and uncertain efficacy (see Discussion), while estimating the longer-term impacts of the interventions on fish stocks would require detailed modelling.

If spatial restrictions were enforced, and in absence of adaptive behaviour by fishers or other changes, compliance would result in a 38% (SE±6%) decrease in mean catch size from 1.84kg to 1.14kg for trap fishers. In contrast, mean catch size for gill net fishers was 17% (SE±4%) larger in non-restricted areas (1.64kg) than proposed restricted areas (1.40kg). Catch sizes for both gear types would decrease if a closed period was enforced during October-November. However, the costs of compliance with a temporal closure differ between gear types; mean catch size for trap fishers would decrease by 4% (SE±1%) from 1.41kg to 1.36kg, whereas gill net fishers would experience a 25% (SE±7%) decrease from 1.53kg to 1.14kg (Table S5.2). If spatial as well as temporal restrictions were implemented and enforced, compliance with both would result in a 21% (SE±3%) decrease in mean catch size (from 1.41kg to 1.11kg) across trap fishers and a 21% (SE±7%) decrease (from 1.53kg to 1.21kg) across gill net fishers (Table S5.2).

These potential changes in catch size would lead to an equivalent change in income received by fishers. Income loss could amount to US$0.58 per day for trap fishers and US$0.35 per day for gill net fishers (Figure 5.6). Compliance with both spatial and temporal closures would lead to a 55% reduction in the number of trips and a 65% reduction in catch for the fishers surveyed (Table S5.2).

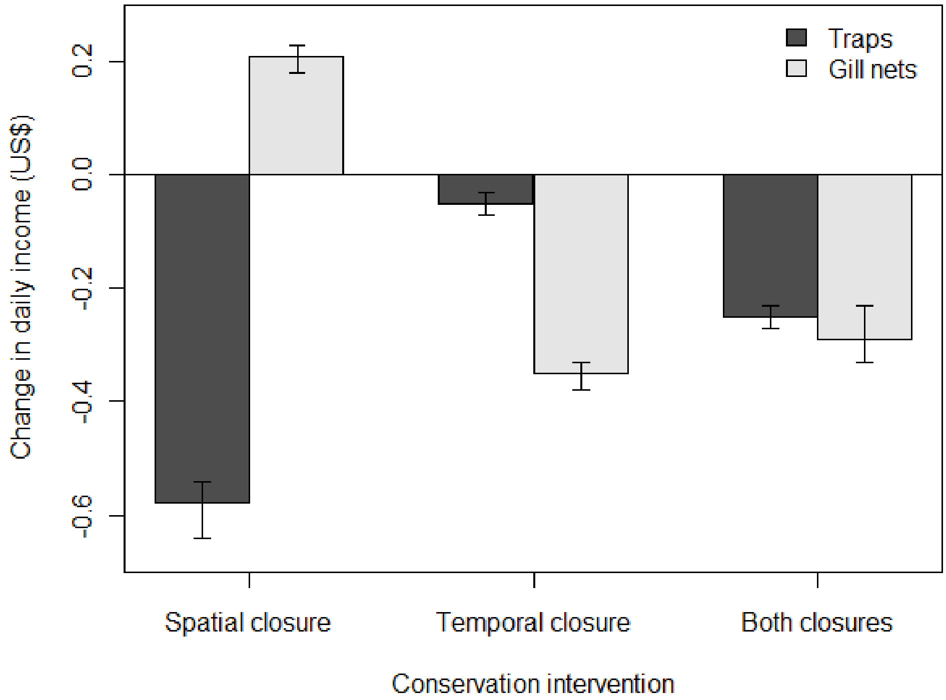


Figure 5.6. Potential impacts of spatial and temporal interventions, as well as both interventions combined, on the daily income of trap fishers and gill net fishers.

5.4 Discussion

Although conservation interventions often have immediate, short-term costs for local people (Naidoo et al., 2006; Daw et al., 2011a), the nature of these costs and how they impact livelihoods is often unclear (Cinner et al., 2008; Holland and Herrera, 2009). I examined the potential impacts for local people of two conservation interventions for Lake Alaotra, spatial and temporal closures, which purport to protect fish where and when they spawn. I found that if the interventions were enforced and complied with they would not only have significant costs to fishers but also impact fisher groups differently.

Accounting for obvious predictors of catch size (such as time spent fishing, number of gear items used, and time of year), the models suggest spatial closures would have little or no adverse effect on gill net fishers but lead initially to smaller catch sizes for trap fishers, reducing their already meagre income by over one third. Conversely, a closed period in October-November would incur greater costs for gill net fishers, reducing their catch and income from fishing by one quarter.

The models confirm that greater fisher effort in terms of amount of gear used and time spent fishing results in larger catches. However, catch size and choice of location are also influenced by gear-specific factors. The planned restricted areas are located in marsh and edge habitat, and trap fishers use these areas primarily because they need to affix their traps to marsh plants. Furthermore, core marsh areas where restrictions aim to protect the Alaotran gentle lemur also appear to be favourable habitat for fish. In contrast, gill net fishers require open areas to avoid entanglement with vegetation, thereby travelling further onto the lake and avoiding restricted areas in the marsh and at the lake edge. This primarily explains why trap fishers are more likely than gill net fishers to be affected by the proposed restricted areas. Additionally, inter-annual differences may be associated with water levels and climate change. Depth measurements were up to 45% lower from 2009 to 2010 and could partly explain the greater inter-annual variation in catches for gill net fishers. Fishers stated that low rainfall and extended cooler temperatures early in 2010 limited fish movement and growth, resulting in reduced stock and ultimately reduced catch sizes.

Compliance with conservation interventions is critical to their effectiveness, but imposing interventions typically generates resentment and high levels of non-compliance (Milner-Gulland and Rowcliffe, 2007; Bunnefeld et al., 2011), and rule-breakers can create short- and long-term costs for legitimate resource users (Gavin et al., 2010). Robust estimates of the costs of compliance can be used to improve management plans and minimise real costs to resource users (Carwardine et al., 2010; Adams et al., 2011). Within the study area, compliance with existing and previous conservation and fishery regulations has generally been very low. My research indicates that high compliance costs may be a primary reason for this, probably in conjunction with a lack of effective enforcement (see Keane et al., 2008).

Scientific uncertainty regarding fish stocks, their spatial distribution, and resource use can lead to uncertainty in managing fisheries (Halpern et al., 2006; Holland and Herrera, 2009). Despite the long-term benefits of restricted areas and active fisheries management, the immediate and short-term costs may be too great for local people to bear and resulting variable or weak compliance undermines conservation targets (Roberts et al., 2001; Shertzer and Prager, 2007; McClanahan et al., 2009). Although economic incentives and perceptions of fairness may influence behaviour, fishers' motivations often reflect convenience, habit, and skill with a particular gear type (Béné and Tewfik, 2001; Sommerville et al., 2010a). Accordingly, differences in costs of conservation for different fisher groups suggest local management efforts need to be targeted to take this variation into account (McClanahan and Mangi, 2004). Because seasonal closures are frequently less biologically effective than spatial closures (Grantham et al., 2008), Lake Alaotra's temporal closure is planned to be phased out in favour of spatial restrictions (R. Lewis, DWCT, pers. comm.). An alternative approach might be to increase flexibility through mobile or dynamic spatial and temporal closures, which could distribute costs and benefits more equitably among fisher groups (see Grantham et al., 2008; Game et al., 2009), and be better received by fishers and ultimately more effective (Berkes, 2003).

While my models estimate potential costs to fishers if planned spatial or temporal interventions were enforced, the full impact of restricting fishing locations may be more or less severe, depending on adaptive changes in fisher behaviour and distribution following implementation. Most studies focus on the ecological consequences of redistributing fishing effort (Dinmore et al., 2003; Hiddink et al., 2006; Greenstreet et al., 2009) and few consider the impacts of spatial or temporal interventions on displaced or resident fishers (Bohnsack, 2000; Charles and Wilson, 2009). Redistribution of effort to non-restricted areas would increase fisher density, potentially leading to further declines in catches for displaced as well as resident fishers, at least over the short term (Cinner et al., 2008; McClanahan, 2010). Conversely, fishers may leave the fishery, attempt to change gears, or increase effort in existing locations or at times not impacted by the intervention, potentially mitigating some effects of the intervention (Cinner et al., 2011; Daw et al., 2012).

Many millions of people depend on artisanal fishing for their livelihood throughout the year (FAO, 2010). My research provides methods to improve understanding of fishers' spatial and temporal behaviour at a scale relevant for conservation planning.

An improved understanding of fisher behaviour can inform fisheries management (see Fulton et al., 2011) and contribute to conservation by ensuring that the potential impacts on resource users can be accounted for and mitigated during the design and planning stages of interventions. My study provides the foundation for further analyses of fishers' spatial behaviour, such as agent-based behavioural modelling to predict how fishers respond and adapt to change.

CHAPTER 6 UNDERSTANDING THE DRIVERS OF FISHER EFFORT AND SPATIAL BEHAVIOUR AT LAKE ALAOTRA, MADAGASCAR

6.1 INTRODUCTION

Inland fisheries are widely recognised as providing a significant amount of food and income for rural communities (Welcomme et al., 2010). However, the resources of most large inland fisheries are frequently fully utilised or overexploited (FAO, 2010; Welcomme, 2011b) and the importance of considering fisher behaviour to improve fisheries management has been emphasised (Wilen et al., 2002; Salas and Gaertner, 2004; Cinner et al., 2008). Understanding the effort and spatial behaviour of fishers should allow fishery managers and conservation planners to minimise the adverse impacts of interventions, increasing the likelihood of fisher support and compliance. However, factors that influence fishers' time investment and spatial behaviour are rarely identified or taken into account when planning and implementing interventions.

To date, most fisheries literature focuses on commercial fishing fleets in developed countries and comparatively little is known about the complexities of subsistence and artisanal fisheries in developing countries (Welcomme et al., 2010). However, the dynamics of artisanal fisheries are similar to those of commercial fisheries in that fisher decision-making and behaviour is influenced by a combination of economic, biological, and social factors, as well as personal preferences (see Abernethy et al., 2007; Tidd et al., 2011). These factors can be grouped into seven categories: tradition and personal preferences (Pálsson and Durrenberger, 1990; Bjarnason and Thorlindsson, 1993), knowledge and information (Gillis et al., 1993), fishing experience and ability (Parker and Sutherland, 1986), risk strategy (Holland and Sutinen, 1999; van Oostenbrugge et al., 2001), practical constraints such as type of vessel and/or gear used (Smith and Zhang, 2007), management constraints such as spatial and/or temporal closures (Dinmore et al., 2003), and stochastic or exogenous constraints such as weather, other human activity, or price of fish (Cinner and McClanahan, 2006).

The theory of the ideal free distribution (IFD) states that harvesters will distribute themselves in relation to resource availability; the number of individual harvesters aggregating at various locations will be proportional to the amount of resources available at each location (Fretwell and Lucas, 1969). The term 'ideal' assumes that harvesters have accurate knowledge of the distribution of targeted resources (such as fish species) and the term 'free' assumes that harvesters are able to move between locations without constraint (Kacelnik et al., 1992; Gillis, 2003). Although the IFD has limitations in practical application to small-scale fisheries, it can provide a useful framework and starting point for understanding fisher behaviour (Abernethy et al., 2007).

Rational choices and utility maximisation are primary tenets of traditional economic theory (Morse, 1997; Güth, 2008), and have frequently been applied within fisheries research to explain fisher decision-making and behaviour (Holland, 2008; Daw et al., 2011b). Microeconomic models assume that fishers have complete knowledge of fishery characteristics and use this to make fishing decisions entirely intended to maximise profit, which is the measure of their personal utility or 'wellbeing' (see Balakrishnan et al., 2000; Edwards-Jones, 2006). In line with recent challenges to this approach by behavioural economists, based on empirical evidence for how decisions are *actually* made rather than how they *should* be made (see Thaler, 2000; Ariely, 2009; Hastie and Dawes, 2010; Kahneman, 2011), it is now increasingly recognised that fishers' strategies or choices can vary considerably and involve a range of compromises that drive their patterns of fishing behaviour (Abernethy et al., 2007; Daw, 2008; Holland, 2008).

This chapter uses information from in-depth semi-structured interviews, focus groups, and catch interviews with fishers to examine factors influencing the amount of time fishers spend fishing and why they fish where they do. If drivers of fisher behaviour are taken into account in conservation planning, it should be possible to develop interventions that are more effective, equitable, and less costly (to fishers), and hence more likely to be supported and complied with by local communities.

6.2 METHODS

6.2.1 Study site

Please refer to Section 3.1 in Chapter 3 for details of the study site.

6.2.2 Data collection

Please refer to Section 3.2 in Chapter 3 for an overview of general data collection methods. Methods specific to this chapter are provided below.

Data were collected in June and July 2009 and also over 14 months from October 2009 to December 2010, comprising two dry seasons (2009 and 2010) and one wet season (2010). I conducted structured catch interviews (n = 1,800), semi-structured background interviews (n = 405), and semi-structured scenario interviews (n = 221) with Anororo-based fishers; further details of interview methods are provided in Section 3.2.4. A total of 784 fishers participated in one or more forms of interview. Respondent codes were assigned to all participating fishers to preserve their anonymity (Bernard, 2002).

A total of 537 individual fishers participated in catch interviews; 248 fishers (46%) were interviewed more than once. Catch interviews were 5 to 10 minutes in duration and comprised a series of questions about the fisher's fishing activity that day, including fishing location (as defined by the fisher), gear type, and effort used. Catch interviews also included counting the number of fish caught and measuring fish lengths (n = 27,064). Total catch weights were estimated using species-specific length-weight relationships (see Chapter 4). A total of 158 fishers participated in catch interviews as well as background interviews. Each background interview collected information on the fisher's demographics, reliance on fishing for livelihood, and fishing behaviour including type of gear and fishing location(s) used. Scenario interviews explored how fishers' spatial distribution had changed over time, particularly whether they had added or dropped locations since the previous interview and the reasons for change. A total of 110 fishers participated in both sets of semi-structured interviews as well as catch interviews.

6.2.3 Data analysis

Data were analysed using R version 2.14.2 (R Development Core Team, 2012). Analysis of fisher behaviour was based on data from background interviews with fishers (n = 405). In order to account for overdispersion I used a negative binomial generalised linear model (GLM) to determine the factors influencing fisher effort, measured as time spent fishing, and identify different groups of fishers. Time spent fishing is the most meaningful measure of fisher effort, both in biological and economic terms, and also allows for comparison between gear types (Abernethy et al., 2007; Daw et al., 2011b). Five socioeconomic characteristics of fishers were assessed as explanatory variables: (1) age category, (2) total number of people supported in the household, (3) level of education (up to primary school or secondary school and above), (4) whether the fisher's household had an alternative livelihood or source of income, and (5) type of gear used for fishing.

Years of fishing experience was significantly and strongly positively correlated with fisher age category ($r = 0.81$, $p < 0.0001$) and therefore not included as an explanatory variable. Almost all fishers interviewed (99%) were from the Sihanaka ethnic group, and 98% had lived in Anororo since birth. Most interviewees (86%) stated that fishing was their primary livelihood, and many of these fishers stated they also had an alternative source of income at times during the calendar year. Lack of variation in ethnicity, whether fishers were native to Anororo, and primary occupation meant it was not possible to examine the influence of these factors on fisher effort and they were therefore not included in the analysis.

Information-theoretic approaches such as Akaike's Information Criterion (AIC; Akaike, 1974) use measures of predictive power to rank models and quantify the magnitude of difference between models (Burnham and Anderson, 2002; Bolker et al., 2009). I used AIC model selection and model averaging to determine model-averaged coefficients. The global model was run using the MASS package in R; the MuMIn package was used for model comparison and averaging. All models where AIC differences were less than three were included in the candidate set of models for model averaging (Burnham and Anderson, 2002).

I analysed data from catch interviews for a total of 1,757 fishing trips by 515 individual fishers; separate analyses were conducted for a sub-sample of 788 fishing trips by 151 individual fishers who had also participated in background interviews. Catch interviews where no fish were measured (n = 39) because fishers had already sold their catch prior to interview, did not want to have their fish measured, or did not have time for their fish to be measured, were excluded from analyses. Interviews with fishers who had fished in the immediate vicinity of the village were infrequent (n = 4) and also excluded from analyses. Anororo-based fishers use a variety of gear types to fish in Lake Alaotra and adjacent marsh. Using data from catch interviews, I compared the distribution of catch and effort to that expected under the ideal free distribution (IFD).

Fisher responses to semi-structured interview questions were categorised into common themes for analysis. Response sample size varies according to whether background or scenario interviews were used to collect the data and because fishers sometimes gave vague or ambivalent responses that could not be categorised explicitly. Data are presented as the percentage of interviewees providing a particular response to questions regarding (a) the reasons for choosing a fishing location, (b) whether their chosen location(s) had changed during the study period and why, and (c) the reasons for using their selected gear type. I summarized differences between fisher groups and compared them using chi-square and Fisher's exact tests.

6.3 RESULTS

6.3.1 Drivers of fishing effort

Results from the negative binomial GLM indicated that gear type and number of people supported were significant predictors of time spent fishing (Table 6.1). Fisher age category, level of education, and presence of an alternative livelihood in the household were not significant explanatory variables. These results confirmed that gear type could be used to categorise fishers at broad scale.

Table 6.1. Results of the negative binomial generalised linear model of fisher profile variables explaining fisher effort measured as time spent fishing. [a, b, c, d] Baseline levels are ‘Age15-24’, ‘primary school education’, ‘no alternative livelihood’, and ‘traps’, respectively. Significant values are in bold.

Explanatory variables	Estimate	SE	*z*	***P***
Intercept	5.9886	0.2931	20.397	<0.0001
Age category[a]				
Age25-34	-0.1793	0.1055	1.697	0.0897
Age35-44	0.0071	0.1074	0.066	0.9471
Age45-54	-0.1756	0.1186	1.478	0.1393
Age55+	0.0203	0.1261	0.161	0.8724
Total supported	0.0406	0.0136	2.975	**0.0029**
Education[b]				
Secondary	0.0832	0.0449	1.852	0.0640
Alternative livelihood[c]				
Yes	0.0342	0.0591	0.578	0.5634
Gear type[d]				
Gill nets	0.7292	0.0481	15.148	**<0.0001**
Cast nets	1.1004	0.1332	8.250	**<0.0001**
Line & hook	1.4453	0.2766	5.218	**<0.0001**
Hand methods	0.3528	0.1581	2.228	**0.0259**

Fishers who had participated in catch interviews as well as background interviews (n = 151) were grouped by gear type to compare characteristics and fishing activity, which showed that the characteristics of fishing activity differ significantly across gear types (Table 6.2). Cast net and line & hook fishers caught the greatest weight of fish per trip, sold the greatest proportion of fish per trip, and spent the greatest amount of time fishing per trip. Gill net and line & hook fishers travelled furthest from Anororo village to fish, while trap fishers and fishers using hand methods fished closest to the village. Fishers using hand methods were least experienced, supported the least number of people per household, and were least likely to have an alternative livelihood. Less-experienced fishers were generally less likely to have an alternative source of income. Fishers using cast nets had greatest fishing experience, supported the greatest number of people per household, and all had an alternative livelihood. Compared to gill net fishers, trap fishers had greater fishing experience, travelled less per fishing trip, supported more people per household, and were more likely to have alternative income.

Table 6.2. Characteristics of fishers and their fishing activity by gear type. Standard errors (SE) are shown in parentheses. ANOVA results refer to differences between gear types for each characteristic (*df* = 4).

Characteristic	**Gear type**					
	Trap	**Gill net**	**Cast net**	**Line & hook**	**Hand methods**	**ANOVA**
Number of fishers in cluster (n = 151)*	88	58	5	4	9	-
Mean catch per trip (kg)	1.66 (±0.14)	1.73 (±0.27)	5.28 (±0.82)	3.34 (±1.19)	0.81 (±0.41)	F=5.74 **p<0.001**
Mean proportion of catch sold per trip	65% (±1.7)	68% (±3.2)	86% (±6.4)	83% (±6.7)	82% (±10.3)	F=2.04 p=0.087
Mean effort (time spent fishing in hours) per trip	1.63 (±0.04)	2.83 (±0.19)	5.32 (±0.40)	6.85 (±1.44)	1.55 (±0.41)	F=89.22 **p<0.0001**
Mean one way distance travelled (km)	3.69 (±0.09)	4.83 (±0.18)	3.77 (±0.23)	4.83 (±1.49)	3.18 (±0.68)	F=9.30 **p<0.0001**
Mean years of fishing experience	18.9 (±0.52)	16.8 (±0.67)	31.3 (±2.44)	17.5 (±7.64)	13.4 (±2.57)	F=7.03 **p<0.0001**
Mean number of people supported in household	4.8 (±0.07)	4.6 (±0.10)	5.3 (±0.42)	3.5 (±1.04)	3.0 (±0.53)	F=4.82 **p<0.001**
Proportion with alternative livelihood	80%	71%	100%	100%	56%	-

* Sums to >151 because 13 fishers used two gear types during the study.

When fishers were grouped by age category to explore links with results for grouping by gear type it was evident that fishers aged 15 to 24, who have least fishing experience, caught and sold a lower amount of fish than older fishers (except those aged 55 or more) and travel furthest from the village to fish (Table S6.1 in Appendix S6). Fishers aged 55 or older, who have most fishing experience, caught and sold the least amount of fish, fished closest to the village, supported the most people per household, and all had an alternative livelihood. Generally, number of people supported and incidence of alternative sources of income increased with age.

6.3.2 Spatial distribution of fisher effort

Across all fishing trips, irrespective of gear type and fisher identification, fishers at Lake Alaotra appear to conform to an ideal free distribution (IFD); the proportion of effort (i.e., time spent fishing) allocated to fishing locations is directly proportional to the proportion of catch derived from those locations (Figure 6.1a). However, combining catches and effort across all fishing trips assumes all fishers have equal knowledge and ability to catch fish, and can select fishing locations freely and without costs or restrictions. Grouping fishers by type of gear used confirms differences in catch:effort relationships between gear types as well as deviations from the predictions of the IFD (Figures 6.1b to 6.1f).

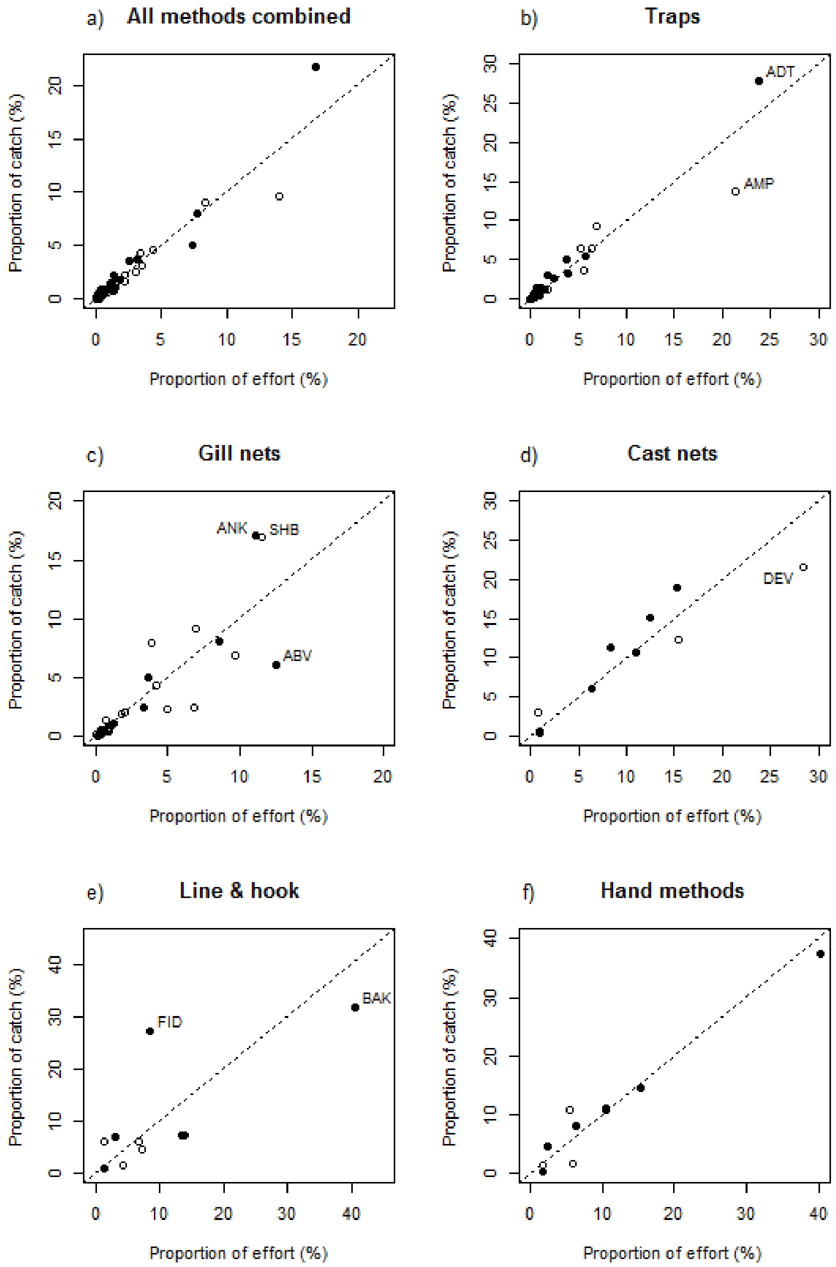


Figure 6.1. Proportions of catch and effort observed at fishing locations in Lake Alaotra over the study period, calculated across all gear types and for each gear type. The two hand methods (*Jinjira*: rod & bubble and *Mangodo*: slap & bubble) are combined. Catch is measured as total weight caught and effort is measured as total number of hours spent fishing at the location. Solid circles represent fishing locations within proposed restricted areas; open circles represent locations within non-restricted areas (see Table 6.3 for characteristics of each labelled location). The dotted line represents the 1:1 prediction of the ideal free distribution.

Heteroscedasticity within the dataset is clear, and considerably more variation occurs at fishing locations where proportions of catch and effort are high (Fligner-Killeen test, $\chi^2 = 24.64$, $df = 6$, $p < 0.001$). There were no explicit patterns or differences between years to explain this deviation from the IFD. Linear models to explore factors influencing deviation for each gear type were inconclusive. However, for each gear type, some general patterns can be drawn from the characteristics of locations that deviate particularly strongly from the IFD (Table 6.3).

Table 6.3. Characteristics of locations with greatest deviation from the ideal free distribution by gear type. [a] Deviation is proportion of catch minus proportion of effort. A positive deviation occurs where proportion of catch exceeds proportion of effort; a negative deviation occurs where proportion of effort exceeds proportion of catch.

Location	**Location characteristics**				
	Index of deviation from IFD[a]	**Distance from village (km)**	**Habitat**	**Restricted area**	**Number of fishers in sample**
Traps					***292***
Andratsilanina (ADT)	4.15	4.1	Edge	Yes	58
Amparihy (AMP)	-7.72	1.3	Marsh	No	66
Gill nets					***155***
Ankororo (ANK)	5.91	3.4	Edge	Yes	24
Sahabe (SHB)	5.41	6.9	Lake	No	30
Ambavasaha (ABV)	-6.43	2.0	Edge	Yes	26
Cast nets					***25***
Deversoir (DEV)	-6.79	4.1	Edge	No	10
Line & hook					***14***
Farihi 'i Daganera (FID)	18.73	5.5	Marsh	Yes	4
Lasin 'i Bakoto (BAK)	-8.56	4.8	Edge	Yes	2

For trap fishers, the two locations deviating most from the IFD were Andratsilanina (which shows higher catch than expected for the proportion of effort allocated to it) and Amparihy (which shows lower catch than expected for proportion of effort allocated), two of the most frequently used fishing locations. Andratsilanina is located within a proposed no-take zone at the lake-marsh edge approximately 4km north of Anororo village and is viewed by many trap fishers as a good fishing location with high fish abundance. In contrast, Amparihy is located within a degraded and open section of the marsh close to the village. Amparihy is also a high-traffic area where collectors (i.e., fish buyers) wait for fishers returning from fishing trips, and local people bathe, do laundry, and bring cattle and other livestock for water. Despite the relatively high level of habitat degradation, fishing, and other activities at Amparihy, fishers continued to fish in this location due to its close proximity to the village, indicating that they value convenience more highly than potentially uncertain catches elsewhere.

For gill net fishers, the two locations with catches most disproportionately higher than predicted by the IFD for proportion of effort expended were Ankororo and Sahabe, which are considerably further from the village than Ambavasaha, the location furthest below the predicted catch:effort relationship. Ankororo and Sahabe are very suitable locations for gill net fishers because they are away from high-traffic areas and vegetation where gill nets could be damaged, entangled, or dislodged by other fishers. Ankororo is perceived by fishers as a good fishing location with relatively high fish abundance, and is located within a proposed no-take zone at the lake-marsh edge. Conversely, although Ambavasaha is also within a proposed no-take zone, it is a very high-traffic area at the lake-marsh edge where the main drainage channel from adjacent rice fields meets the lake. Collectors gather at Ambavasaha to buy catches from fishers and often do laundry while waiting. As a result Ambavasaha is a heavily used and degraded area with poor water quality; it is also relatively shallow, which is unfavourable for standard-size gill nets (~80cm high). Despite these shortcomings, gill net fishers continue to fish at Ambavasaha due to its relatively close proximity to Anororo village and potential for good catches, albeit based on past rather than recent performance. Disproportionate fishing activity for returns at Ambavasaha also suggests that Anororo-based gill-net fishers are risk averse. These fishers often stated that, despite smaller catches, fishing at the edge of the lake was safer than travelling further offshore.

For cast net fishers, Deversoir is a popular location when using relatively small mesh sizes (~1cm) and targeting small fish. Because Deversoir is a good nursery area for fish, cast net fishers often caught hundreds of juvenile fish during a single day there. However, although catches were large in terms of number of fish, Deversoir falls below the IFD line because of the considerable amount of time required to catch a large number of juvenile fish and the corresponding low total weight of fish caught. Despite a relatively low return on time invested, these fishers were reluctant to change their routine or manner of fishing.

For line & hook fishers, Farihi 'i Daganera had a higher catch than predicted by the IFD for effort expended. It is an open fishing location within the proposed strict conservation zone of the marsh, surrounded by high quality marsh habitat and sheltered from extreme weather. It is also ideal habitat for blotched snakehead, the species targeted most by line & hook fishers. In contrast, Lasin 'i Bakoto fell furthest below the IFD prediction for line & hook fishers despite also being within a proposed no-take zone. It is located on the lake-marsh edge and open to strong winds, waves, and currents. However, compared to Farihi 'i Daganera, Lasin 'i Bakoto is relatively close to the village, easier to get to, and has lower fishing density. These factors probably attracted line & hook fishers to this location.

Proportions of catch and effort were also calculated and compared to the IFD for the sub-sample of fishers (n = 151) who had participated in background interviews and thereby provided additional data about their fishing activity (Figure 6.2). The pattern observed for the sub-sample was very similar to that for the larger sample of all fishers (n = 515) participating in catch interviews. This suggests that choices of fishing location in the sub-sample of fishers completing background interviews were representative of all fishers completing catch interviews. The sample of fishers participating in background interviews (n = 405) was therefore used to further explore the drivers of fisher behaviour.

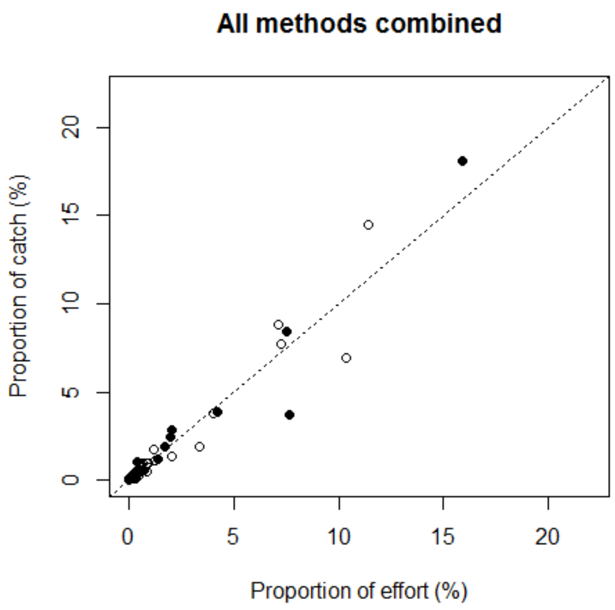


Figure 6.2. Proportions of catch and effort observed across all gear types at fishing locations in Lake Alaotra, calculated for all fishers who also participated in background interviews (n = 151). Catch is measured as total weight caught and effort is measured as total number of hours spent fishing at the location. Solid circles represent fishing locations within proposed restricted areas; open circles represent locations within non-restricted areas. The dotted line represents the 1:1 prediction of the ideal free distribution.

6.3.3 Factors influencing fishers' spatial behaviour

Although fishers participating in catch interviews used a single location per fishing trip, fishing locations are used adaptively according to changing conditions over the year; fishers use a median of two locations over the calendar year (see Chapter 4). Almost all fishers participating in background interviews (98% of 403) stated that they have continued to fish at the same location(s) over the last five years. Routine was the most frequently cited reason for a fisher's decision to use a location, followed by catch defined as good catches and high fish abundance (Table 6.4). Being familiar with or having good knowledge of a fishing location was also relatively important and is linked to routine; however, only seven of these fishers stated that a long family history of fishing at a location was a reason for using a fishing location.

Table 6.4. Reasons provided by fishers (n = 403) for choosing fishing locations. The number and proportion of fishers stating each reason are grouped by gear type.

Reason	Traps (n = 213)		Gill nets (n = 167)		Cast nets (n = 10)		Line & hook (n = 7)		Hand methods (n = 6)		All gear types (n = 403)	
	n	%	n	%	n	%	n	%	n	%	n	%[a]
Routine – Usual location. Always uses this or these locations.	157	73.7	134	80.2	10	100.0	5	71.4	2	33.3	309	76.7
Catch – Many fish are present, good catches, good quality fish and/or presence of specific target species.	43	20.2	38	22.8	1	10.0	2	28.6	1	16.7	85	21.1
Familiarity – Fisher has good knowledge of the location (e.g., how to get there, move around the location, and catch fish) and the location is appropriate for the fisher's skills and ability. May have a long history of fishing there.	39	18.3	14	8.4	0	0.0	1	14.3	0	0.0	54	13.4
Suitability – Location has characteristics (e.g., water level or habitat) that suit the fisher's gear type or manner of fishing (e.g., camps out). There are favourable environmental characteristics for fishing; calm (no wind), sheltered or protected, location can be used all year. Fisher preference.	22	10.3	16	9.6	4	40.0	0	0.0	3	50.0	46	11.4
Travel – Close to village or rice field. Allows time for other activities. Close to collectors who buy fish. Location is not clogged with invasive plants. Ease of travel, accessible.	22	10.3	8	4.8	0	0.0	1	14.3	3	50.0	34	8.4
Fishers – No or few thieves. No large seine nets that destroy gear. Camaraderie, enjoyable because friends fish there.	10	4.7	3	1.8	0	0.0	0	0.0	0	0.0	14	3.5

[a] Proportions sum to >100% because 129 respondents (32%) nominated multiple reasons.

The vast majority (94%) of fishers interviewed used traps or gill nets and reasons for selecting fishing locations differed significantly for these fishers (Fisher's exact test, $p = 0.024$). Specifically, a significantly greater proportion of trap fishers nominated familiarity and travel time as a reason for selecting a location (chi-square tests: familiarity $\chi^2 = 11.79$, $df = 1$, $p < 0.001$; travel time $\chi^2 = 6.53$, $df = 1$, $p = 0.011$). All cast net fishers stated that routine was a reason for selecting their fishing location. Reasons why fishers selected a fishing location did not differ between age categories (Fisher's exact test, $p = 0.597$).

Despite the stated high degree of consistency in relation to location choice during background interviews (Table 6.4; routine), 81% of fishers subsequently participating in scenario interviews (n = 221) stated that they were using a different set of locations since the background interview. The most frequently cited reasons for fishers being pushed out of or pulled into different fishing locations were related to catch size, travel, and water level (Table 6.5). In most cases, factors that pushed fishers out of their preferred fishing location(s) were more powerful, which indicates their reluctance to move otherwise and is consistent with initial responses specifying routine as the main driver of location choice. Only travel-related factors pulled rather than pushed a higher proportion of fishers to a new location, which was always closer to Anororo village. Reasons for switching locations did not differ between gear types or between age categories (Fisher's exact tests, $p = 0.742$ and $p = 0.420$, respectively).

Table 6.5. Reasons provided by fishers (n = 178) for being pushed out of or pulled into other fishing locations. The number and proportion of fishers stating each reason are grouped for pushed and pulled.

Reason	**Pushed**		**Pulled**		**Total**	
	n	**%**	**n**	**%**	**n**	**%**[a]
Catch – Catch and fish size. Pushed out of location due to poor catches or small fish size. Pulled into other locations for better catches or larger fish. Follow seasonal movement of fish; fishers follow fish movement to continue to have a catch.	49	27.5	17	9.6	66	37.1
Travel – Pushed out of previous locations because access became difficult due to invasive plants. Pulled in because of proximity to village, residence, or rice field, less travel time, or allowing time for other activities. May change seasonally or with second season rice cultivation activities in the marsh, due to age or health of fisher, with changes in personal circumstances, and/or may involve risk aversion.	12	6.7	24	13.5	36	20.2
Water level – Pushed out of locations due to unusually low seasonal water levels in 2010.	31	17.4	0	0	31	17.4
Fishers – Pushed out because of overcrowding or presence of thieves, or due to presence of methods that make it difficult to use their preferred gear. Pulled in because location is uncrowded or less crowded, has fewer thieves, or recommended by other fishers.	16	9.0	14	7.9	30	16.9
Suitability – Pushed out because location characteristics change over time and become unsuitable for preferred fishing strategy. Pulled in because location characteristics are better suited to the fisher's choice of gear or manner of fishing (e.g., camps out).	12	6.7	9	5.1	21	11.8
Other – Pushed out due to habitat degradation and/or poor water quality (e.g., invasive plants degrading fishing locations in the marsh, dirty or stinking water). Pulled in to trial or explore additional fishing location(s).	6	3.4	2	1.1	8	4.5

[a] Proportions sum to >100% because 14 respondents (8%) nominated multiple reasons.

A fisher's choice of fishing location is influenced, and often constrained, by the type of gear used (e.g., traps cannot be used in open lake habitat). Only 4% of fishers participating in background interviews (n = 405) had changed their gear type in the previous five years. This was primarily due to theft or destruction of gear (e.g., by fire) or practical constraints such as the fisher's age and/or fitness. Across all fishers who had not changed type of gear used (n = 388), the most frequently cited reasons for choice of gear type were routine, competence, and expense (Table 6.6). Most fishers (80%) specified routine and/or competence as the primary reasons for continuing to use their preferred type of gear.

However, the proportion of fishers nominating each reason for gear selection varied significantly according to type of gear selected (Fisher's exact test, $p < 0.0001$). A significantly smaller proportion of gill net fishers considered affordability a motivating factor in gear choice (chi-square test, $\chi^2 = 63.1$, $df = 4$, $p < 0.0001$), reflecting the relatively greater initial and ongoing expense of gill nets compared to other methods. The most important reasons for gear choice were affordability for fishers using line & hook, competence for fishers using hand methods, and routine for those using gill nets, traps, or cast nets. Of fishers using traps or gill nets (94% of the sample), trap fishers were significantly more likely to state competence or low expense as reasons for gear choice (chi-square tests: competence $\chi^2 = 4.85$, $df = 1$, $p = 0.0276$; inexpensive $\chi^2 = 15.13$, $df = 1$, $p = 0.0001$).

Only gill net fishers stated that they selected their gear type because it did not contravene regulations (i.e., "gill nets respect regulations"; n = 5). Gill net mesh sizes reported during catch interviews were usually larger (mean 33mm) than those reported for traps (mean 25mm) or cast nets (mean 18mm). Whereas gill net mesh sizes often exceeded the 40mm minimum specified by the Regional Fisheries By-laws, trap and cast net mesh sizes were frequently smaller than legally required (40mm and 35mm, respectively). Gill net fishers who stated 'respect for regulations' as a reason for gear choice tended to be fishers aged over 35 with a mean of 22 years fishing experience and who wanted to continue being fishers in the future. They also valued the marsh as habitat for fish and other wildlife, and viewed the regulations as good for protecting the fishery and their livelihood. The line & hook method, which targets blotched snakehead, is the only gear type legally permitted for use during the closed period to allow fishers to catch fish for food. Blotched snakehead is a carnivorous fish that feeds on juvenile tilapia; in an attempt to control its abundance there are no regulations regarding catches of snakehead. None of the interviewed line & hook fishers stated 'respect for regulations' as a driver of gear choice.

Type of gear used differed significantly across age categories for trap and gill net fishers (Fisher's exact test, p = 0.009). Specifically, a significantly greater proportion of fishers aged over 44 (n = 88) used traps (67%) rather than gill nets (33%) (chi-square test, χ^2 = 10.23, df = 1, p = 0.0014); there was no difference in the proportion of fishers aged under 44 (n = 292) using traps (53%) rather than gill nets (47%) (chi-square test, χ^2 = 0.877, df = 1, p = 0.349). While a significantly greater proportion of fishers using hand methods (*Jinjira*: rod & bubble and *Mangodo*: slap & bubble) were aged up to 24 (chi-square test, χ^2 = 95.35, df = 4, p < 0.0001) and fishers using cast nets were aged over 44 (chi-square test, χ^2 = 71.536, df = 4, p < 0.0001), these results are based on small sample sizes. The reasons why fishers used a gear type did not differ between age categories (Fisher's exact test, p = 0.383).

Table 6.6. Reasons provided by fishers (n = 388) for selecting their primary type of gear used. The sample comprises fishers who had not changed gear type over the previous five years. The number and proportion of fishers stating each reason are grouped by gear type.

Reason	**Traps (n = 200)**		**Gill nets (n = 163)**		**Cast nets (n = 10)**		**Line & hook (n = 7)**		**Hand methods (n = 7)**		**All gear types (n = 388)**	
	n	**%**	**n**	**%**	**n**	**%**	**n**	**%**	**n**	**%**	**n**	**%**[a]
Routine – habit, preference, desire to use, enjoyment.	96	48.0	108	66.3	6	60.0	1	14.3	1	14.3	212	54.6
Competence – fisher ability, skill, knowledge, easy to use, not tiring, practical.	66	33.0	43	26.4	2	20.0	1	14.3	3	42.9	116	29.9
Inexpensive – few expenses, no money for other methods, no other equipment.	27	13.5	5	3.1	3	30.0	4	57.1	2	28.6	41	10.6
Flexible – does not take much time, allows free time to do other activities, easy to move from one location to the next, easy to transport.	16	8.0	10	6.1	0	0.0	0	0.0	0	0.0	26	6.7
Fishers – tradition, recommended by friends, gear type has fewer thieves than others.	13	6.5	6	3.7	2	20.0	3	42.9	1	14.3	25	6.4
Other – work alone, not easy to get materials for other methods, gear type is respectful of regulations.	2	1.0	10	6.1	0	0.0	0	0.0	0	0.0	12	3.1

[a] Proportions sum to >100% because 44 respondents (11%) nominated two reasons.

6.4 DISCUSSION

Examining the drivers of fisher effort and behaviour within Lake Alaotra's artisanal fishery, this study shows that fishers conform generally to the ideal free distribution (IFD). Although Anororo-based fishers are mainly people of habit, they distribute throughout the lake and marsh to locations where they believe they will have good or consistent (but not necessarily maximal) catches. Departures from the IFD are primarily tied to convenience for fishers and/or the relatively unique environmental or anthropogenic characteristics of a fishing site, which often assume importance for gear-specific reasons. Fishers invest time in fishing to a degree they consider appropriate to achieve a sufficient catch, also taking into account their broader sets of personal, social, and economic interests.

Fisheries research at all scales has historically focused on accumulating knowledge of fishery resources, including species distributions, trends in fish stocks, harvest rates, and habitat characteristics, while affording comparatively little attention to fishers' effort-related and spatial behaviour (Salas and Gaertner, 2004; Daw et al., 2011b). As a result, many instances of ineffective fishery management can probably be attributed more to misunderstandings about fisher behaviour, preferences, and decision-making processes than to poor knowledge about target species or habitat dynamics (Hilborn, 1985; Daw, 2008; Holland, 2008). Although fisher behaviour must be considered in order to understand fishery dynamics (Salas and Gaertner, 2004), my results show that the drivers of this behaviour may vary according to environmental, gear-use, and motivational factors. These factors can differ between sites and therefore analysis should occur at local level.

Type of gear used is the major factor influencing both fishing effort and choice of fishing location for Anororo-based fishers. Gear- or method-based variation in fisher effort and spatial distribution has often been reported as characterising smaller-scale subsistence or artisanal fisheries to a greater extent than larger commercial fisheries, particularly in developing tropical countries (Abernethy et al., 2007; Daw et al., 2011b; Hamerlynck et al., 2011). These fisheries tend to be highly heterogeneous in fisher demography, fish species, gear used, environmental features, and catch sizes (Teh et al., 2007; Cinner et al., 2009b), and may be only part of a suite of livelihoods that households engage in (Allison and Ellis, 2001; Cinner and Bodin, 2010). The broad range of fisher objectives within these fisheries usually means that time spent fishing is valued by opportunity cost (Daw et al., 2011b), which in this study was also tied to type of gear used. This suggests that grouping fishers by gear type may provide the most informative classification when analysing behaviour within multi-gear artisanal fisheries, especially in terms of spatial behaviour.

It was not unexpected that fishing effort would differ across gear types, given that environmental and habitat factors may constrain the type(s) of gear that a fisher can use, or render some methods more suitable than others (see Welcomme, 2001). Whereas variation in gear use often occurs to target different species (see Gillis et al., 1993; Abernethy et al., 2007), for Anororo-based fishers this variation is mediated by differences in habitat within their fishing arena. For example, gills nets require relatively large open areas that are clear of obstructions, which are typically further from the village and require more travel time to access. Conversely, traps cannot be used in open lake habitat because they need to be affixed to supporting materials such as reeds, which occur in marsh and lake edge habitats that are closer to the village and typically require less travel time to access. Similarly, fishers using hand methods require shallow water and/or marsh habitat, which both occur closer to the village. Fishers using cast nets usually made numerous (i.e., 100+) net casts along routes over one to two kilometres adjacent to lake or marsh edges in order to catch an adequate quantity of fish for food and income, which entailed a large investment in time spent fishing. Additionally, fishing is most favourable for gill net and cast net fishers when weather conditions are calm or they are sheltered from wind. These gear-specific influences of environmental and habitat factors on fisher effort also mediated fishers' choices of fishing location (and therefore spatial distribution), suggesting that gear type is particularly important for understanding fisher behaviour within multi-habitat fisheries.

While fishers were best grouped for effort on the basis of gear type, and to a lesser extent the number of people they supported, there were indirect but specific ties to fisher age. Broadly, the youngest fishers invested high levels of effort by travelling furthest from the village to fish. This could be because suitable closer locations are occupied by established fishers, so that 'junior' fishers must venture further from the village to find vacant fishing sites or wait for closer locations to become available. Conversely, the eldest fishers invested relatively low effort by fishing closest to the village, perhaps because they (i) had earliest access to closer fishing sites, (ii) rank highest on fishers' social hierarchy, (iii) are more likely to be involved in alternative income-producing activities, and/or (iv) wish to spend more time with their family, even though the result that they support more people than younger fishers suggests they should invest greater time fishing. These interpretations are consistent with the findings of studies focusing on the role of social norms and relationships in artisanal fisheries (see Ruttan, 1998; Cinner et al., 2005; Coulthard, 2008), indicating that classifications based on gear type can also be heterogeneous and a nuanced understanding of behaviour is always required.

The finding that Anororo-based fishers conform generally to the IFD is in contrast to results reported by Abernethy et al. (2007) for artisanal Anguillan reef fishers, where departures from the IFD were linked to fisher age, experience, and target species, as well as type of gear used. However, in my study species were not explicitly targeted, fisher age or experience were not drivers of catch (see Chapter 5), and catch was an important factor for fishers when selecting or changing their fishing location(s). This indicates that fishers pursued rational strategies or objectives, to the extent that their knowledge permitted, when distributing their effort spatially, which is usually more characteristic of larger-scale commercial fisheries than smaller-scale artisanal systems (Branch et al., 2006; Powers and Abeare, 2009). Despite this, and in contrast to the central assumption of the IFD that all individuals aim to optimise profits (see Gillis, 2003; Abernethy et al., 2007), fishers did not fish in ways to maximise returns. Rather, in line with equivalent findings by Béné and Tewfik (2001), Cabrera and Defeo (2001), Salas and Gaertner (2004), and Daw (2008), fishers' decisions on effort distribution were mediated by trade-offs with their other interests.

Trade-offs made by fishers were in terms of convenience, routine, gear usability and maintenance, or predictability of catch. The strategies preferred by many fishers were (a) spend more time fishing closer to home rather than invest that time travelling to distant locations or (b) continue fishing in a familiar location where catches are more predictable but sometimes small. Swain and Wade (2003) reported similar strategies for fishers of snow crab in the Gulf of St. Lawrence, as did Daw (2008) for lobster fishers in Nicaragua, indicating that a broad range of fishers compromise or satisfice instead of optimising utility (Simon, 1955; Foxon, 2006) and this is not only because they are boundedly rational (Jones, 1999; Gintis, 2007).

Trade-offs differed according to type of gear used such that the departures from the IFD were explained best at gear level. Many gill-net fishers appeared to be markedly risk averse, preferring the relative safety and consistency of smaller catches close to the lake edge over the risks of travelling further onto the lake for potentially larger returns, especially when weather conditions were poor or uncertain. Cast net fishers were particularly driven by routine, which is consistent with their habit of repeatedly using a few pre-baited fishing routes (widely acknowledged as theirs by other fishers) despite the potential for larger catches by varying routes or trialling new routes. Many line and hook fishers were primarily influenced by whether a location was physically comfortable to fish at for lengthy fishing trips, and whether it was easy to access.

The predominance of routine as a driving factor for Anororo-based fishers extends from choice of fishing location through to selection of gear and persistence with that type of gear. Specifically, fishers' decisions about location and gear intertwine and, once choices are made, persistence is bound by considerations of (i) familiarity with relevant site characteristics, (ii) competence with the type of gear used, and (iii) the perceived costs of changing location and/or gear. Provided catches remain adequate for the amount of effort invested, which depends on trade-offs with fishers' interests other than catch, there is considerable inertia within the fishery and a reluctance to change.

Routine, habit, and/or familiarity with location or gear are increasingly identified as key drivers of fisher behaviour in studies of fishing decisions. Models developed by Bockstael and Opaluch (1983) to explain responses to uncertainty by New England fishers indicated that selection of location was driven by past choices, which reflect the perceived costs of changing location as well as habit or traditions. Holland and Sutinen (2000) added site familiarity as a major factor in location selection by New

England trawl fishers, whereby a fisher's experience at a site, and lack of experience at other sites, influenced their expectations of profitable returns from a location and therefore their spatial behaviour. Eggert and Tveteras (2004) found that gear use by Swedish fishers depended on past choices, cost, and routine, and was highly inert; the same factors drove urchin divers' location choices in California, regularly overriding evidence of potentially larger catches at other, but unfamiliar, locations (Smith, 2005). These findings align with results in my study to suggest that the inertia arising from routine, familiarity, and the costs of converting to different fishing gear or locations may underpin fisher spatial behaviour, and these drivers may override heterogeneity in perceived risks and returns.

In contrast to other studies (for example, see Bockstael and Opaluch, 1983; Holland and Sutinen, 2000; Christensen and Raakjær, 2006) family traditions in location or gear use were relatively unimportant as drivers of spatial behaviour and gear choices for Anororo-based fishers. Similarly, age was not influential in choice of gear except for older or relatively young fishers. Many fishers over 44 years of age preferred to use traps because they could be used closer to the village; cast net fishers were often older because of the greater experience required to use this method. Hand methods (rod & bubble and slap & bubble) are favoured by many fishers aged up to 24 years who are not invested in fishing throughout the year but fish late in dry season when water level is lowest and these methods yield good returns for fishing effort. This is consistent with the result that most hand-method fishers nominated competence and affordability as key reasons for selecting the method, with less emphasis on routine; hand methods require less competence and expense than other methods and are most suitable for itinerant fishers. Familiarity with fishing location was most important for trap fishers because they must be selective about where they place their traps; travel and opportunity costs may operate in tandem with familiarity to mediate choice of fishing location (Daw, 2008). The high importance of competence for fishers' choice of gear suggests that they perceive benefits in catch from greater expertise in using a particular type of gear, reiterating that catch ultimately sits alongside routine, costs, and convenience as a key driver of fisher spatial behaviour.

Catch size and travel costs, which interlinked with water level and access to fishing locations, were most likely to motivate fishers to change their spatial behaviour. Poor catches and/or high travel costs were most likely to push changes in fishers' spatial behaviour. In contrast, and as discussed above, the prospect of better catches and/or lower costs *per se* had less effect on changing fishers' spatial behaviour because of the constraining influences of routine and familiarity. These patterns of driver impact show that although Anororo-based fishers conformed generally to the IFD and made rational spatial decisions, they are risk averse and suffice with sub-maximal catches under conditions of uncertainty. A key management implication of this is that these and similar fishers may be less responsive to purely-economic incentives to modify their behaviour than has historically been expected (see Holland, 2008). Identifying and understanding the drivers of fishers' spatial behaviour is therefore essential to manage fisheries effectively.

This study provides further evidence that fishers do not conform to the predictions of traditional economic theory; rather, in line with findings from behavioural economics, their strategies and choices may vary considerably and involve various compromises. The principles of behavioural economics also account for the inertia or reluctance of Anororo-based fishers to change gear or fishing locations but persist with consistent sub-optimal catches. Fishers are boundedly rational, like all economic agents, and fish under uncertainty (see Tversky and Kahneman, 1974; Kahneman, 2003). Uncertainty is particularly characteristic of artisanal fisheries in developing countries, largely due to the absence of advanced fishing technology; fishers depend on experience, shared traditional knowledge, and heuristics (i.e., 'rules of thumb') to access a resource that is hidden from view (Holland, 2008; Daw et al., 2011b; Wise et al., 2012). Decisions made under uncertainty tend to be risk averse and therefore maintain the *status quo*, as predicted by prospect theory (see Kahneman and Tversky, 1979; Kahneman, 2011), particularly when tradeoffs with other interests are involved (Holland, 2008; Milner-Gulland, 2012). Risk aversion for fishers in this study related primarily to uncertain catches rather than gear theft and destruction, contravening gear-size regulations, or personal safety (except for gill-net fishers deterred by uncertain weather), in contrast to studies suggesting that these factors can drive fisher behaviour (Abernethy et al., 2007; Daw et al., 2011b).

The study confirms the utility of the IFD as a tool for analysing the spatial behaviour of fishers as resource users. Although fishers lacked perfect knowledge and did not always maximise profit, catch size was sufficiently important to influence behaviour and fishers generally made reasonably informed choices based on past experience or knowledge of costs; the assumptions of the IFD were adequately approximated to uphold it (see Prince and Hilborn, 1998; Gillis, 2003; Branch et al., 2006). Relatively high levels of conformity to the IFD confirmed commonality in fishing strategies at broad scale. Departures from the IFD were not only due to bounded rationality but primarily to gear- and location-specific factors as well as a range of trade-offs that fishers individually considered important. Variation in spatial behaviour was greatest where catch and effort were highest, indicating that there may be greater scope for trade-offs or spatially-specific effects as catch and effort increase; higher return for effort invested may allow fishers more options or flexibility in their behaviour.

From a management perspective it is pragmatic to understand and account for fisher behaviour collectively (Béné and Tewfik, 2001; Holland, 2008; Cinner et al., 2010). The results of this study suggest that trade-offs and variation in spatial behaviour in multi-gear artisanal fisheries may be best understood by grouping fishers by the type of gear they use. Failure to understand the diversity of interests and motivations that influence fishers' spatial behaviour might not only lead to poor prediction of their responses to change and interventions, but also ineffective fishery management.

CHAPTER 7 ARTISANAL FISHERS' PERCEPTIONS OF MANAGEMENT INTERVENTIONS: USING SCENARIOS TO UNDERSTAND HOW FISHERS WOULD RESPOND TO CHANGE

7.1 INTRODUCTION

Rural communities in developing countries frequently rely on the resources of inland fisheries for food and income (SEAFDEC, 2005; Welcomme et al., 2010; Hamerlynck et al., 2011). However, these resources are often overexploited and managed under uncertainty (FAO, 2010), which may include a lack of understanding about fisher behaviour and how it could be taken into account in designing management actions (Wilen et al., 2002; Salas and Gaertner, 2004; Cinner et al., 2008). While restrictions on fishing areas and/or effort are often implemented to manage fisheries, their consequences are typically assessed in ecological terms (Hiddink et al., 2006) even though it is equally or more important to evaluate impacts, or perceived impacts, on fishers (Hilborn, 1985; Béné and Tewfik, 2001; Branch et al., 2006). Because fishing restrictions depend on compliance by fishers for efficacy, it will be highly informative to know in advance how fishers would change their behaviour in response to interventions. Compliance with conservation interventions in turn largely depends on fishers and other resource users having good knowledge of regulations and reasons for them, as well as positive attitudes towards management (Keane et al., 2011).

Fishers within small-scale artisanal fisheries perceive and respond to management interventions in a diverse range of ways (Salas and Gaertner, 2004; Gelcich et al., 2005; McClanahan et al., 2005). Although responses to interventions may be partly tied to heterogeneity in fishing objectives and factors influencing fishing behaviour (see Chapter 6), they often depend greatly on fishers' attitudes and perceptions about the impacts of interventions, or management practices generally (McClanahan et al., 2005). A comprehensive understanding of resource-users' perceptions of, attitudes toward, and probable responses to an intervention can not only inform management decision-making when developing policy (Allison and Ellis, 2001; Smith et al., 2005) but also increase the legitimacy of an intervention and provide a basis for evaluating its impacts and effectiveness (Gillingham and Lee, 1999; Weladji et al., 2003). The perceptions and responses of resource users can also be site and context dependent (Daw et al., 2012), and hence must be understood in order to render an intervention locally appropriate. Furthermore, adaptive responses that appear to be appropriate at an individual level in the short-term can often have adverse impacts on the broader social-ecological system; for example, fishers may increase effort or fish within new areas in response to reduced catches, thereby accelerating stock declines and undermining system resilience (Gunderson et al., 1995; Wilson, 2006; Cinner et al., 2011). Failure to anticipate the responses of resource users to interventions and changing conditions may be one of the main causes of many management failures (Berkes, 2003; Polasky, 2008; Mills et al., 2010; Weeks et al., 2010b).

This chapter draws upon data from in-depth semi-structured interviews and focus group sessions with fishers in Lake Alaotra, Madagascar, to understand fishers' perceptions of potential impacts of regulations on their fishing behaviour, including fishers' suggestions for better management. I then use scenario analysis to show how fishers are likely to adjust their behaviour in response to restrictions and changing conditions, such as spatial or temporal interventions, greater fisher density, and changes in catch size. Because benefits in terms of fish abundance and catch are unlikely to be realised over the short-term following intervention (Halpern and Warner, 2002; Russ et al., 2005; Cucherousset et al., 2007; Suski and Cooke, 2007), immediate costs to fishers will greatly influence the success of management actions during implementation and on an ongoing basis. Knowledge of fishers' perceptions of interventions and probable responses to change can improve conservation planning by allowing interventions to be developed that have fewer costs to fishers and greater social acceptability, and are therefore more likely to be complied with (Salas and Gaertner, 2004; Adams et al., 2011).

7.2 METHODS

7.2.1 Study site

Please refer to Section 3.1 in Chapter 3 for details of the study site.

7.2.2 Data collection

Please refer to Section 3.2 in Chapter 3 for an overview of general data collection methods. Methods specific to this chapter are provided below.

Management interventions deemed in this chapter to be current at Lake Alaotra include the planned series of no-take zones around the edge of the lake (which are primarily ‘on paper’ rather than implemented and enforced), designated protected areas within the marsh, the annual closed period from 15 November to 15 January, as well as gear, mesh size, and minimum fish size restrictions (see Sections 3.1.6 and 3.1.7 in Chapter 3, and Section 4.3.3 and Table 4.5 in Chapter 4). It should be noted that the label ‘current’ does not necessarily imply acknowledgement by fishers, actual implementation, or widespread practice. As a point of difference to these current interventions, the scenario situations explored in this chapter are for hypothetical circumstances similar to or arising from interventions.

If resource users, such as fishers, perceive researchers to be closely linked with or informants for management authorities they may be concerned about hidden agendas and whether the information they provide will be used in ways detrimental to them. To receive meaningful assistance and honest answers from fishers it is imperative to have their trust and confidence (Jacobsen et al., 2012), particularly when sensitive or potentially contentious issues may be addressed (see Briggs, 1986; Esterberg, 2002).

Accordingly, all members of the research team made conscious and consistent efforts throughout the study to build credibility with fishers, gain and maintain their trust, and respect confidentiality at all times. I re-emphasised that the study was a student project, reassured fishers that we were not policing their activities, and ensured their anonymity in data records by assigning codes to respondents instead of using names (Bernard, 2002). The research team frequently explained to fishers that only honest and complete answers to interview questions could produce effective suggestions or solutions for managing the fishery in ways that take fishers’ interests and concerns into account. The large number of fishers electing to participate in the research (i.e., approximately 70% of the estimated 1,100 fishers residing in Anororo), their candid and pragmatic answers to (often sensitive) questions, and their numerous unsolicited insights indicated that we had the trust and confidence of the fishers we worked with.

Self-reported measures, such as direct interview questioning, can be a source of social desirability bias (i.e., systematic error that arises from the desire of respondents to provide the answer he/she believes the researcher wants to hear), which should be anticipated and minimised when using interview methods and evaluating responses (Fisher, 1993; King and Bruner, 2000; Randall and Koppenhaver, 2004). Although indirect techniques and self-administered surveys have been used to prevent or reduce social desirability bias, they carry other sets of errors and might not provide accurate estimates of respondents' attitudes and behaviours (Fisher and Tellis, 1998; St. John et al., 2010).

Singer et al. (1995) found that assurances of anonymity typically increase response rate and reduce social desirability bias for sensitive interview questions, while Jones et al. (2008) found that interviewing people independently and on multiple occasions maximised the power of interviews. These strategies were employed throughout the study to address potential social desirability bias. Additionally, some questions posed during semi-structured interviews overlapped intentionally to revisit topics that were potentially sensitive and facilitate probing for further details.

Semi-structured background interviews (n = 405) were conducted with Anororo-based fishers during November and December 2009. Each background interview collected information about the fisher's demographics, reliance on fishing for livelihood, fishing behaviour, attitudes and perceptions of the impacts and/or benefits of fishing regulations, and suggestions for managing the fishery. Semi-structured scenario interviews (n = 221) were conducted 11 to 12 months after the background interviews (i.e., November and December 2010) and only with fishers who had participated in background interviews. This facilitated corroboration of data as well as assessment of changes over time. Scenario interviews explored (a) how fishers would be affected by and respond to spatial and temporal restrictions on their fishing activity, including how their fishing effort or spatial distribution may change, as well as (b) impacts and responses to changing fishing conditions such as increased fisher density and increasing or decreasing catches. Scenarios are frequently used in similar research to explore capacity to anticipate change and adapt (see Cinner et al., 2011). Focus group sessions were used to triangulate data from individual interviews and provide additional contextual information.

7.2.3 Data analysis

7.2.3.1 Interviews

Fishers' responses to interview questions were categorised into common themes. Response sample sizes vary according to whether background or scenario interviews were used to collect the data and because fishers sometimes gave vague or ambivalent responses that could not be categorised explicitly. Perceived percent changes in catch size or income were calculated based on fisher responses, which included estimates of the change in the number of fish they would catch, the fullness of a bucket, or the expected cash value of catch before and after a scenario. I summarized differences between groups of fishers and compared them using chi-square and Fisher's exact tests. Data were analysed using R version 2.14.2 (R Development Core Team, 2012).

7.2.3.1.1 Fisher knowledge and perceptions of current management interventions

Fisher knowledge and perceptions of management interventions that either currently existed or were in the planning stages were explored during background and scenario interviews. Fishers were first asked whether regulations existed, and then asked questions to assess their knowledge of the spatial, temporal, and gear-related regulations. Each fisher was subsequently asked for their perceptions about the regulations, including whether the regulations were good or bad for them.

7.2.3.1.2 Perceived impacts of, and adaptive responses to, management interventions

To examine the perceived impacts and adaptive responses to spatial and temporal interventions, fishers were asked how they would respond if their fishing location(s) became a restricted area, how this change would affect them, and also how the closed period affects them. Interview questions were open-ended; depending on the fisher's response, a series of follow-up questions were asked to probe for further details about their responses, such as where they would fish and if they would change the type of gear they used.

7.2.3.1.3 Adaptive responses to fishing intensity

Fishing intensity was defined as the number of fishers using a fishing location. Many fishers stated during informal discussions that overcrowded locations and competition between fishers was a problem, and felt that decreasing the space available for fishing would exacerbate this issue. I hypothesised that if spatial restrictions were implemented and fishers changed location in response, this would increase fisher density, and therefore fishing intensity, within the receiving non-restricted areas. In order to understand how this potential response to spatial interventions may impact fishers over the short-term, fishers were asked during scenario interviews whether their catches would change if there were half the number of fishers, twice as many fishers, or four times as many fishers using their fishing location(s) compared to current numbers, and what they would do in response to this change. Interview questions were open-ended, allowing fishers freedom to provide their own views rather than selecting from lists. Responses were categorised into three common themes, following the approach used by Cinner et al. (2011); however, in my study the categorisation refers to change in fishing intensity rather than impacts on system resources and resilience:

1) **Continue**: continuing to fish as usual, without adapting;
2) **Amplifying**: adapt in a manner that would increase fishing activity, such as changing gear, location, or tactics in order to increase catch, increasing effort, or guarding their equipment or location to ensure catches were not reduced; or
3) **Dampening**: adapt in a manner that would decrease fishing activity, including reducing effort and not doing anything else (which fishers often referred to as '*kamo*'), reducing effort and doing something else for income, or stopping fishing altogether. Although the word '*kamo*' translates directly as 'lazy', it does not carry negative connotations and can suggest that not doing something else for income is a conscious choice; other activities such as spending time with family or resting may be assigned greater priority if the benefits of fishing are reduced.

Fishers were asked to provide both short-term (after one month) and long-term (after one year) responses; however, the types of short- and long-term responses were not significantly different for any scenarios (Fisher's exact tests, $p > 0.06$) and therefore only short-term responses are reported. The effects of fishers' responses on level of fishing activity were then examined, as well as the potential impacts of this for the fishery.

7.2.3.1.4 Fishers' suggestions for management

Fishers were asked a series of questions during both sets of interviews to obtain their opinions about how to best manage the fishery, including their recommendations for the size and locations of no-take zones, and the timing and duration for a closed period. Again interview questions were open-ended, which allowed fishers freedom to provide their own views rather than selecting from lists. Responses were categorised into common themes and presented as the percentage of fishers providing each response.

7.3 RESULTS

7.3.1 Fisher knowledge and perceptions of current management interventions

Of 405 fishers participating in background interviews, 372 (92%) were aware that regulations existed to control activities in the lake and marsh. The majority of these fishers (93%) correctly believed that responsibility for administering and enforcing these regulations was held by one or more of the following organisations: Durrell Wildlife Conservation Trust, the Department of Water and Forests, or the Fisheries Service. Only 15% of fishers identified the Federation of Fishers as having an active role in management; less than 5% mentioned that transfers of management to local communities existed, were underway, or were being planned (see Section 3.1.5 in Chapter 3 for further details of governance).

7.3.1.1 Spatial interventions

Almost two thirds of fishers completing background interviews and also subsequent scenario interviews (n = 221) were aware that spatial interventions either currently existed or had been implemented in the past but without success or enforcement (Table 7.1). Although no fisher could name all of the fishing locations within the strict conservation zone in the centre of the marsh or no-take zones around the lake edge, fishers knew the most popular fishing locations nearest the village that were affected by the current (i.e., ‘on paper’ but not implemented or enforced) spatial interventions. However, of the 38 fishing locations nominated by the fishers, half were incorrectly listed as restricted areas. A few fishers (2%) correctly stated that hunting lemurs or burning reeds is prohibited in the marsh; however, they incorrectly believed that fishing was permitted throughout the marsh. These results indicate that Anororo-based fishers have incomplete knowledge of spatial restrictions.

When asked to explain the positioning of restricted areas, 92% of fishers listing marsh locations (n = 26) correctly stated that the reason was to protect habitat for the Alaotran gentle lemur and other wildlife. Across the 139 fishers aware of restricted areas, the most frequently cited explanations for no-take zones around the lake edge were to protect habitat favourable for fish (48%) or the spawning grounds of fish (38%), which matches management objectives; the other responses were: i) skilled fisheries technicians determined the location of no-take zones (6%), ii) the zones are locations for stock enhancement (2%), or iii) the fisher did not know or the response was ambiguous (6%).

Compared to fishers who fished outside restricted areas, a greater proportion of fishers who fished within restricted areas stated that there were no restricted areas (28% versus 42%, respectively), though this difference was not statistically significant (chi-square test, $\chi^2 = 3.41$, $df = 1$, $p = 0.065$). Some fishers who indicated there weren’t any spatial restrictions did so because they believed the interventions were only ‘on paper’ and not implemented or enforced. Fishers identified good as well as bad aspects of restricted areas, but mostly good (Table 7.1).

Table 7.1. Fisher awareness and views on restricted area regulations. The number and proportion of fishers who responded are provided for each question.

Questions; scenario interviews (n = 221)		**n**	**%**
Are there restricted areas?	No	82	37
	Yes	139	63
Are there good aspects about the restricted areas?	No	1	1
	Yes	138	99
What are the good aspects? (n = 138)			
1. Increased production and quantity of fish		92	67
2. Protection of fish		32	23
3. Protection of wildlife for tourism		11	8
4. Protection of habitat / environment		3	2
Are there bad aspects about the restricted areas?	No	131	94
	Yes	8	6
What are the bad aspects? (n = 8)			
1. Too large, decreased space available for fishers, lake edge and marsh are prime fishing locations especially in bad weather		4	-
2. Buoys/markers stress fish		1	-
3. Restricted activities in marsh (no rice cultivation)		2	-
4. Conservation zone in marsh causes proliferation of water hyacinth		1	-

7.3.1.2 Temporal interventions

The majority (99%) of fishers interviewed (n = 221) were aware that a closed period existed (Table 7.2) and most of these (98%) accurately identified its current timing from 15 November to 15 January. All fishers aware of the closed period stated that it is intended to protect and increase the production of fish, and thereby improve the fishery and maintain fisher livelihoods, which matches management objectives. Of the 221 fishers interviewed, 81% correctly stated that fishers were permitted to fish for food during the closed period; however, only three fishers (2%) correctly added that line & hook was the only authorised fishing method during the closed period (and none of the three fishers were line & hook fishers). The majority (72%) of fishers aware of the closed period (n = 219), stated they continue to fish during the period (Table 7.2), however, views about the closed period did not differ significantly between fishers who continued to fish and those who did not (Fisher's exact tests, $p > 0.06$). Fishers identified both good and bad aspects of the closed period (Table 7.2).

Table 7.2. Fisher awareness and views on closed period regulations. The number and proportion of fishers who responded are provided for each question.

Questions; scenario interviews (n = 221)		**n**	**%**
Is there a closed period?	No	2	1
	Yes	219	99
Do you fish during the closed period?	No	62	28
	Yes	157	72
Are there good aspects about the closed period?	No	10	5
	Yes	209	95
What are the good aspects? (n = 209)			
1. Increased production and quantity of fish		161	77
2. Protection of fish		25	12
3. Increased catch size / income when fishery is re-opened		21	10
4. Other: prohibited methods not used, time to make more equipment		2	1
Are there bad aspects about the closed period?	No	172	79
	Yes	47	21
What are the bad aspects? (n = 47)			
1. Lack of income / work		31	-
2. Lack of compliance		4	-
3. Lower fish prices		4	-
4. Too long		2	-
5. Eliminates savings		2	-
6. Poor fishing conditions (catch small fish)		2	-
7. Other: change in diet, predatory fish attack small fish		2	-

7.3.1.3 Gear restrictions

Ninety percent of fishers interviewed (n = 405) stated that gear restrictions existed and collectively correctly nominated all 14 fishing methods prohibited under the Regional Fisheries By-laws (Dina de Pêche, 2006). The ‘*Ramangaoka*’ (a one-kilometre seine net with small mesh size), ‘*Valatany*’ (a mud enclosure), and ‘*Sitra*’ (a dip net made from a mosquito net), were the methods most frequently identified as prohibited, by 85%, 41%, and 41% of fishers respectively. Only 44% of fishers aware of gear restrictions (n = 365) stated that small mesh size is also prohibited for legal fishing methods; 70% of these fishers specified a minimum legal mesh size, which in almost all cases (96%) was smaller than that specified in the Regional Fisheries By-laws (which are 40mm for traps and gill nets and 35mm for cast nets). Only four fishers stated correctly the minimum legal mesh size for the type of gear they use.

In only seven of 405 cases (<2%) was a fisher's stated fishing method an illegal method, and these were hand methods and the mud enclosure. Although in five of these cases the fisher was aware of regulations restricting fishing activities, none of the fishers reported their fishing method as one of the methods they nominated as prohibited.

7.3.1.4 Perceptions of management interventions generally

Of fishers aware of management interventions, the majority held a positive future outlook towards restricted areas (97% of 138 fishers) and the closed period (95% of 219 fishers), perceiving future benefits such as increasing quantities of fish, greater income, a thriving fishery, and preservation of resources for future generations. Fishers viewing these interventions negatively perceived future problems from non-compliance, which would render the restrictions ineffective, as well as conflict between authorities and fishers. Fishers aware of both interventions (n = 219) were significantly more likely to envisage bad aspects about the closed period than spatial restrictions (Fisher's exact test, p = 0.0004). While most fishers supported fishing regulations, 19%, who were generally older and had a median of 18 years fishing experience, stated that the regulations are poorly enforced, do not correspond to prevailing circumstances, and fail to account for the needs of fishers. Similarly, 41% of 97 fishers perceiving regulations as bad during background interviews nominated corruption and lack of enforcement as the reasons for this. Fishers of all age categories and gear types expressed the view that a lack of enforcement undermines compliance with regulations, to the detriment of fishers who respect regulations.

7.3.2 Perceived impacts of spatial and temporal interventions

7.3.2.1 Impact on fishing location and effort

When fishers (n = 221) were asked what they would do in response to their fishing location(s) becoming a restricted area, 91% stated they would change location. Of the fishers who specified an alternative location (n = 187), 58% nominated locations a median of 4.6km further from the village than their original location(s), 41% nominated locations a median of 3.9km closer to the village, and 1% nominated a location equidistant to their current location. This indicates that fishers who usually fish closer to the village would shift their location by approximately the same distance as those who usually fish further from the village, and the two groups would essentially swap spatial distributions; this could be termed a 'grass is greener' effect. This trend further illustrates that Anororo-based fishers generally conform to the ideal free distribution (IFD) across fishing locations (see Chapter 6). Of 16 fishers leaving the fishery, 13 stated they would work in rice cultivation, two would grow vegetables, and one would become a fish collector.

Ninety-seven percent of the 157 fishers who stated they continue to fish during the closed period (see Table 7.2) stated that they fish less than they normally do. This is because i) they are afraid of being caught or feel they are stealing (72%), ii) they are only fishing for food (18%), iii) they try to adhere to the regulations but still need to earn money (5%), iv) it coincides with low water level and it is therefore difficult to fish (2%), and/or v) there are no collectors to sell to (1%). Only 18 fishers (11%) stated they change locations during the closed period; 10 of these select locations closer to the village and/or good for hiding from authorities. Sixty-eight percent of fishers stating they fish during the closed period also admitted to selling fish, which is prohibited. Fish were sold to bicycle collectors, covert collectors, within the village market, or secretly from house to house (called '*paraky*', which literally means 'tobacco' to disguise what is being sold). Of the fishers who claimed not to fish during the closed period (n = 62), 58% stated that they worked in rice cultivation over this time; other activities included growing vegetables, making and/or repairing fishing equipment, working in transport, or remaining unemployed.

All fishers taking on alternative work due to interventions stated that the reason for their choice was to earn money and there was no other feasible option. These shifts in livelihood activity, primarily to rice cultivation, may increase pressure to convert marsh habitat into rice fields.

7.3.2.2 Impact on income and daily activities

Most fishers stated that their income would change as a consequence of either spatial or temporal interventions (Table 7.3). The majority of fishers expected a decrease in income due to spatial interventions (median 50% decrease; range 18% to 100%); the perceived decrease in income did not differ significantly across gear types (Kruskal-Wallis test, $\chi^2 = 2.236$, $df = 4$, $p = 0.693$). The majority (78%) of these fishers stated that the decrease would be caused by unfamiliarity with the new location and reduced catch sizes. Trap fishers and gill net fishers stated that 2 days to 3 months (median 7 days for both gear types) would be required to become familiar with a new fishing location; fishers using other gear types stated that familiarisation would require one to two weeks. Fishers would also need to move their fishing equipment to a new location. Of 105 trap fishers interviewed, 69% stated it would be difficult to move all of their traps, while 31% stated it would be easy. In contrast, 30% of 79 gill net fishers interviewed stated it would be difficult to move their nets while 70% stated it would be easy. Only 8 fishers (4%) stated they would change or add a type of gear to their routine.

Of 183 fishers stating that a closed period would change their overall annual income, 31% expected their income would decrease (median 33% decrease; range 5% to 80%). Although gill net fishers expected a greater decrease (median 40%; range 8% to 80%) in income than trap fishers (median 31%; range 7% to 75%), differences were not significant (Kruskal-Wallis test, $\chi^2 = 1.709$, $df = 4$, $p = 0.789$). The majority (82%) of fishers stated that the decrease would be caused by decreased fishing activity and reduced catch sizes, leading to little or no income from fishing over the two-month period. Of the 127 fishers stating that their income for the year would increase, 94% perceived that the closed period would increase the quantity of fish available in the lake and they would catch many fish when the fishery re-opened.

Most fishers stated that either spatial or temporal interventions would cause a change in their daily routine and the majority stated it would be an unwelcome change (Table 7.3). Fishers considered it bad to change routine because their health would be compromised and they would become sickly or less fit, lack sleep and become tired, be stressed, or have no income or reduced income.

Table 7.3. Perceived impacts of spatial and temporal interventions. The number and proportion of fishers who responded are provided for each question.

Questions; scenario interviews (n = 221)		**Spatial**		**Temporal**	
		n	**%**	**n**	**%**
Would the intervention cause a change in income?	No	23	10	38	17
	Yes	198	90	183	83
If yes, would it increase or decrease?	Increase	70	35	127	69
	Decrease	128	65	56	31
Would the intervention cause a change in your daily routine?	No	55	25	16	7
	Yes	166	75	205	93
If yes, is this good or bad?	Good	46	28	94	46
	Bad	120	72	111	54
Would the intervention change the time you have available for other activities?	No	53	24	9	4
	Yes	168	76	212	96
If yes, would it increase or decrease?	Increase	51	30	190	90
	Decrease	117	70	22	10

Interventions would also change the amount of time fishers have available for other activities (Table 7.3); however, the direction of change differed for the two types of interventions. Under spatial interventions, fishers would spend more time doing fishing-related activities such as moving their equipment, becoming familiar with the new location, or spending time guarding their equipment, and would have less time for other activities. The two activities most frequently cited to be reduced were equipment repairs (n = 60; 51%) and time to rest (n = 39; 33%). Only 10 fishers stated they would have less time for other work (e.g., work in rice fields, transport). In contrast, under temporal interventions fishers would be unemployed and have more time for these activities. Relative to the effects of spatial interventions, proportionally more fishers stated that a change in routine caused by the closed period would be good by allowing more rest, a vacation from fishing (i.e., time to recuperate and regain strength), and time for other activities such as work in rice fields. Proportionally more fishers perceiving the change in routine as good either owned their own rice field (51%) or had an alternative livelihood (79%), compared to those who perceived the change as bad (43% and 68%, respectively); however, differences were not statistically significant (chi-square test, $\chi^2 = 2.653$, $df = 1$, $p = 0.103$).

7.3.3 Fishing intensity

Fishers perceived that the number of Anororo-based fishers has increased over time; 99% of fishers stated during background interviews (n = 405) that there were more fishers now than when they began fishing. Almost all (98%) fishers with more than five years of fishing experience (n = 332) perceived that fisher numbers increased substantially from 2005 to 2009, by a median factor of 2 (range 1.1 to 10.0).

During scenario interviews (n = 221), 66% of fishers stated that the current intensity of fishing affects fishing adversely because i) it stresses fish and stunts their growth (51%), ii) fish learn to avoid capture (22%), iii) it reduces fish stock (19%), and iv) fish leave the area (8%). Most fishers (61%) stated that the current intensity of fishing was bad for them. This was due to overcrowding, interference between gear types, competition, the presence of thieves, and disturbance and noise that caused fish to scatter and resulted in small catches. However, perhaps surprisingly, 35% of fishers considered the current fishing intensity good for them. Benefits included being able to work together and discuss fishing techniques, safety in numbers, cooperating against thieves, and being able to surround fish and prevent them escaping. Only 2% of fishers perceived both benefits and costs from the current fishing intensity and 1% considered it neither good nor bad.

7.3.3.1 Adaptive responses to fishing intensity

Of 195 fishers who stated their daily catch size would change if there were half the number of fishers, 98% expected their catch would increase (median increase 60%; range 10% to 700%). In response to this, 80% of these fishers would increase effort and 20% would continue fishing as usual. Fishers increasing effort would add equipment, spend more time fishing, reposition their equipment and spread it out, or improve or better-maintain their equipment. Of the fishers who would continue fishing as usual (n = 36), seven stated they would spend the additional money earned on rice cultivation or other household needs while three stated they would keep the increased catch a secret from others.

The majority of fishers indicated an amplifying response to a decrease in catch size due to greater fishing intensity at their fishing location(s); 68% of 204 fishers for a two-fold increase in intensity and 60% of 216 fishers for a four-fold increase (Figure 7.1). The proportion of fishers adopting each type of response (continue, amplifying, or dampening) did not differ significantly over the short-term (one month) compared to the longer-term (one year) for either two-fold or four-fold increases in fishing intensity (Fisher's exact tests: p = 0.06, p = 0.61). However, for approximately 10% of fishers there was a shift from continuing as usual or amplifying responses to dampening responses if greater fishing intensity persisted over the longer term.

Responses to a two-fold compared to four-fold increase in fishing intensity differed significantly (Fisher's exact test, p = 0.0002) and only 28% of 221 fishers provided the same response to both scenarios (Table 7.4). The four responses with greatest change in frequency between scenarios were: (1) continue as usual, (2) increase effort, (3) change location, and (4) stop fishing. Frequency of the first two of these responses decreased substantially from a two-fold to four-fold increase in intensity, whereas the latter two increased substantially. Generally, fishers' responses shifted from no response to continue as usual, an amplifying response, and then a dampening response (see Table 7.4). Many fishers stated they would trial a different amplifying response before reducing their effort or leaving the fishery. Importantly, 39 fishers (18%) stated they would stop fishing with a four-fold increase in fishing intensity; 24 of these fishers perceived their catches would decrease by 90% or more and 15 stated they would work in rice cultivation as an alternative livelihood if they stopped fishing.

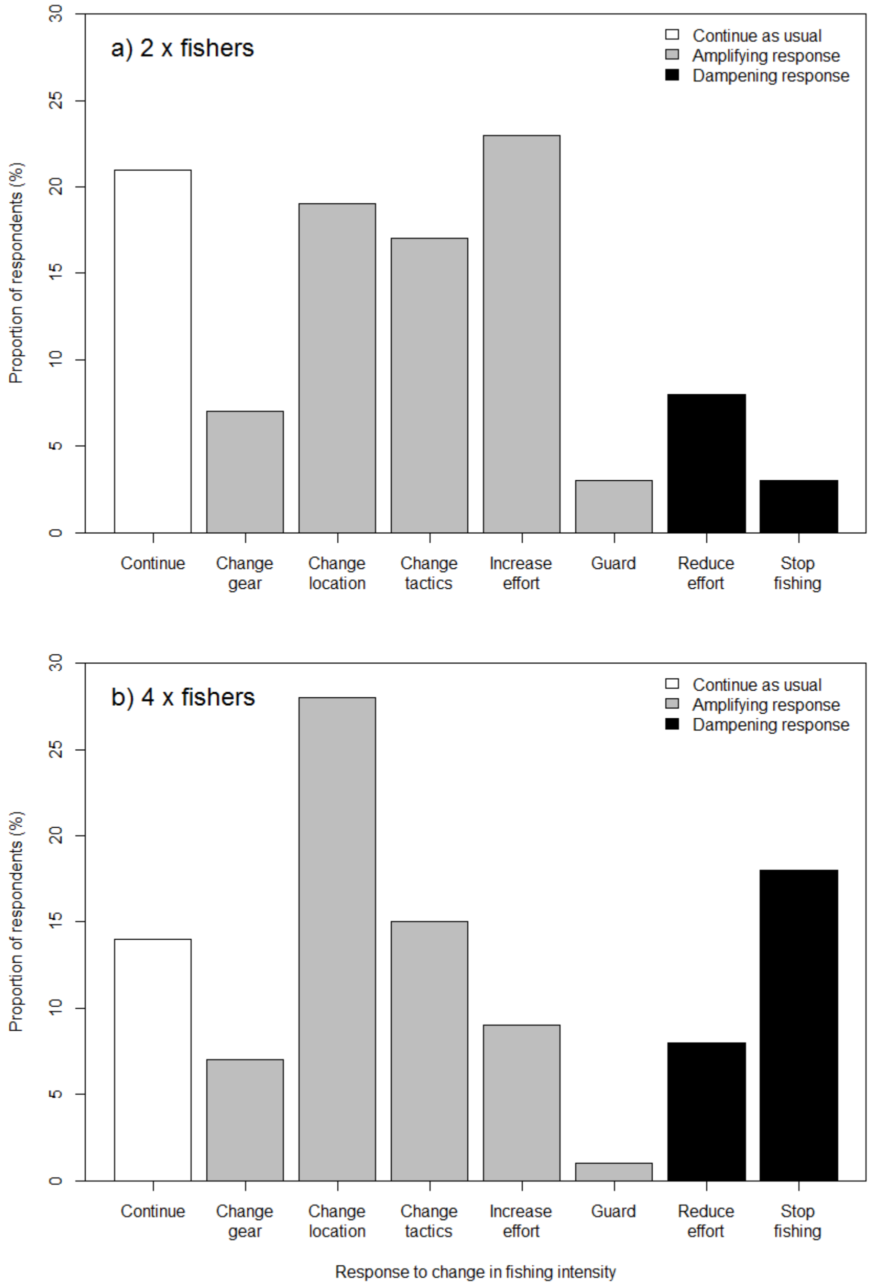


Figure 7.1. Fisher responses to a hypothetical change in fishing intensity; a) a two-fold increase and b) a four-fold increase in the number of fishers using a fishing location. a) n = 204 fishers. b) n = 216 fishers.

Table 7.4. Contingency table showing the number of fishers providing each response to a two-fold increase in fishing intensity (rows) compared to four-fold increase in fishing intensity (columns). Shaded cells refer to fishers providing the same response to both scenarios. Sums in bold are the four responses with greatest change between the two scenarios; arrows adjacent to these responses indicate the direction of change, decrease (↓) or increase (↑), from the two-fold scenario (rows) to the four-fold scenario (columns).

		Due to a four-fold increase in number of fishers, catch would:										
		Not change	Increase	Decrease	Decrease	Decrease	Decrease	Decrease	Decrease	Decrease	Decrease	
Due to a two-fold increase in number of fishers, catch would:	**Fisher Response**	**Catch not change**	**Catch increase**	**Continue ↓**	**Change gear**	**Change location ↑**	**Change tactics**	**Guard**	**Increase effort ↓**	**Reduce effort**	**Stop fishing ↑**	**SUM**
Not change	**Catch not change**	3		2		8			1		1	15
Increase	**Catch increase**			1		1						2
Decrease	**Continue**	1		11	2	6	1	1	2	5	14	**43**
Decrease	**Change gear**			4	2	3	1		1		3	14
Decrease	**Change location**	1		4	4	18	2		1	1	7	**38**
Decrease	**Change tactics**			3	4	9	10		4	2	2	34
Decrease	**Guard**				1	2		1	1	1		6
Decrease	**Increase effort**			2	1	7	16	1	9	8	2	**46**
Decrease	**Reduce effort**			2	1	6	3			1	3	16
Decrease	**Stop fishing**										7	**7**
	SUM	5	0	**29**	15	**60**	33	3	**19**	18	**39**	221

↓ Responses where proportions decreased from the two-fold scenario (rows) to the four-fold scenario (columns).
↑ Responses where proportions increased from the two-fold scenario (rows) to the four-fold scenario (columns).

Under the scenario of a two-fold increase in fishing intensity, 204 fishers (92%) expected their catch would decrease (median decrease 50%; range 10% to 100%). Similarly, under a four-fold increase 216 fishers (98%) expected their catch would decrease (median decrease 83%; range 25% to 100%). Relationships between perceived reductions in catch and fishers' responses are shown in Figure 7.2. With a two-fold increase in fishing intensity fishers would reduce effort or stop fishing only when the reduction in catch exceeded 20%. However, the proportion of fishers stating a dampening response did not increase with greater perceived reduction in catch (Figure 7.2a). This suggests that a fishers' decision to fish or how much effort they use is not sensitive to expected reduction in catch from a two-fold increase in fishing intensity. In contrast, under the four-fold increase scenario fishers stated a dampening response only when expected reduction in catch exceeded 40% (Figure 7.2b). This is because fishers who stated their catch would not change, or only change marginally, under the two-fold scenario expected their catch would change or change substantially under the four-fold scenario. Effectively, fishers jumped up one position on the scale of perceived reduction in catch, so that the proportion of fishers stating a dampening response under the four-fold scenario increased with greater catch reduction.

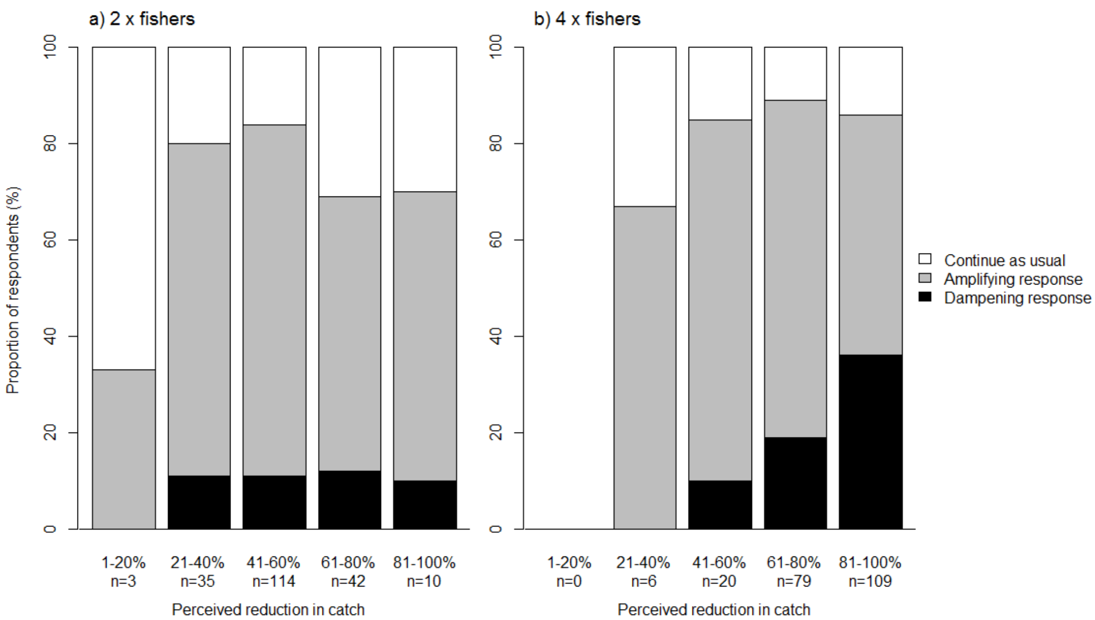


Figure 7.2. Fisher responses to a perceived reduction in catch caused by a hypothetical change in fishing intensity; a) a two-fold increase and b) a four-fold increase in the number of fishers using a fishing location. No fisher stated less than a 21% decrease in catch with a four-fold increase in the number of fishers.

Because fisher behaviour can differ according to type of gear used (see Chapter 6), relationships between fishers' responses to the two fishing intensity scenarios and primary gear used were also explored (Figure 7.3). For trap, gill net, and cast net fishers, dampening responses generally increased with increased fishing intensity. Of the five line & hook fishers in the sample, two (both aged 55+) stated they would fish less with a two-fold increase in fishing intensity; although differences in response frequency between scenarios were not significant (Fisher's exact tests, p = 0.52), the longer the increase in fishing intensity persisted the more these fishers stated they would change to an amplifying response. All line & hook fishers stated an amplifying response to a four-fold increase in fishing intensity. Four of five fishers using hand methods were aged 15 to 24 years, and not deterred by increased fishing intensity and the prospect of reduced catch sizes. Analysis of relationships between fishers' responses to the two fishing intensity scenarios and fisher age category (see Figure S7.1.1) showed that dampening responses generally increased with fisher age, which was probably tied to physical fitness and greater prevalence of an alternative livelihood (see Chapter 6).

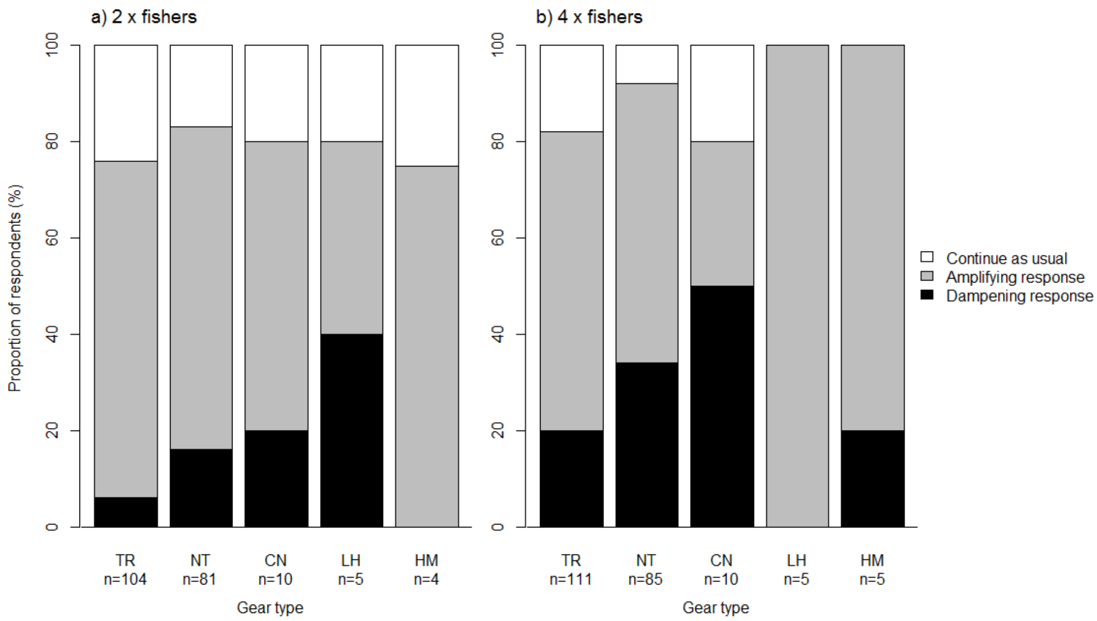


Figure 7.3. Fisher responses by gear type to a hypothetical change in fishing intensity; a) a two-fold increase and b) a four-fold increase in the number of fishers using a fishing location. TR = traps; NT = gill nets; CN = cast nets; LH = line & hook; HM = hand methods.

7.3.4 Fishers' suggestions for management

To help inform management priorities, fishers were asked about the uses and value of the marsh from their perspective. Fishers participating in background interviews (n = 405) stated they use the marsh for a variety of reasons, including materials (96%), fishing (76%), and medicine (53%), and that these were the primary reasons why the marsh had value for them. Fishers generally perceived that managing the marsh is inherent in managing the fishery. Most fishers (87%) made suggestions about how to improve management of the fishery (Table 7.5) and 72% of these fishers suggested enforcing regulations, or specifically, preventing use of destructive fishing methods.

In line with results for fishers' perceptions of gear restrictions (see Section 7.3.1.3), the fishing methods considered most destructive and in need of elimination were large seine nets, mud enclosures, and dip nets, which each catch juvenile fish. Large seine nets were perceived as the most destructive of these methods, in terms of damage to habitat and also damage to or complete loss of other fishers' gear. Although dip nets are legal for catching mosquitofish (*Gambusia holbrooki*) and Madagascar rainbowfish (*Rheocles alaotrensis*), fishers often use them to catch juvenile tilapia (*Oreochromis sp.*) when mosquitofish and rainbowfish are not in season. Because up to hundreds of juvenile tilapia can be caught with dip nets per fishing trip, many fishers believed this method was particularly damaging for the fishery and should be banned entirely. Perceptions of destructive gear were corroborated in focus groups and with informal discussions with fishers.

Of 221 fishers participating in scenario interviews, 96% suggested where restricted areas could be placed best to protect spawning fish, and therefore fish stocks. Fishers nominated 40 locations, the majority of which (55%) are at the lake edge, 38% are in the marsh, and 7% are in the lake. Fifty-one percent of the nominated locations are within planned restricted areas. The ten most frequently suggested locations are all on the lake edge (Table 7.6).

Table 7.5. Fishers' suggestions to improve management of the fishery and the number and proportion of fishers providing each suggestion (n = 352).

Management action suggested	**n**	**%[a]**
Enforcement – General enforcement of regulations for all fishers with no discrimination and no corruption. Responses in this category included statements such as 'authorities should make site visits', 'authorities need to be monitored', and 'punish people who break the rules'.	156	44.3
Stop the use of destructive methods – Enforcement of regulations to stop prohibited methods such as large seine nets, reed walls, mud enclosures, and equipment with small mesh size, as well as stop the use of dip nets, which are frequently used to catch juvenile fish.	135	38.4
Closed period – Adjust timing and/or length of the closed period; ensure closed period is specific to Lake Alaotra and is enforced.	80	22.7
Monitoring – Establish an organisation for ongoing monitoring of fishing activity in the lake and marsh; recruit local fishers to monitor and enforce regulations.	55	15.6
Remove invasive aquatic plants – Eliminate/control invasive plants; re-open channels; renew fishing locations.	48	13.6
Stock enhancement – Introduce new fish species; maintain previous stock enhancement program.	29	8.2
Education – Increase local awareness of management issues to promote compliance with regulations.	19	5.4
Subsidies – Provide subsidies to fishers; alternative income during the closed period; fishing equipment.	15	4.3
Restricted areas – Establish restricted areas to protect spawning fish.	10	2.8
Other – Other suggestions were tree planting to prevent soil erosion; removing mud from the lake; eliminating the blotched snakehead because they eat other fish; stabilising the price of fish; and designating part of the marsh for rice cultivation.	17	4.8

[a] Proportions sum to >100% because 155 respondents (44%) provided multiple suggestions.

Table 7.6. The ten locations most frequently suggested by fishers for restricted areas and the number and proportion of fishers suggesting each location (n = 213). All locations are on the lake-marsh edge and are shown in Figure 4.5 in Chapter 4.

Suggested locations for restricted areas	Within a planned restricted area	n	%[a]
Andratsilanina	Yes	87	40.8
Ankororo	Yes	45	21.1
Ambavasaha	Yes	38	17.8
Tanjombe	Yes	36	16.9
Vohitrandriana	Yes	32	15.0
Bararata miala namana	No	30	14.1
Anosimbandro	Yes	20	9.4
Saha logis	No	17	8.0
Lozokas – natural inlets around the lake edge	-	16	7.5
Lake edge – generally anywhere on the lake edge	-	15	7.0

[a] Proportions sum to >100% because 126 respondents (59%) provided multiple suggestions.

Fisher-nominated considerations to take into account when selecting restricted areas focused on location characteristics and the biological requirements of fish; only 16% of fishers stated that restricted areas should be in locations less frequented by fishers and few (5%) recommended consulting with fishers during the selection process (Table 7.7). Each of the ten most frequently suggested locations for restricted areas had the five location characteristics most often nominated by fishers as important to protect fish (see Table 7.7). The overlap between fisher-nominated locations and the planned restricted areas suggests these areas are appropriately placed to protect fish and fish habitat. However, the median size of planned no-take zones at the lake edge is 2.6km^2, which most fishers (91%) stated was too large. Fishers recommended a median size of 0.04km^2 (range from 50m^2 to 2km^2) for each no-take zone.

Table 7.7. Considerations and location characteristics that fishers suggested should be taken into account when selecting restricted areas to protect spawning fish. Number and proportion of fishers suggesting each consideration or location characteristic are provided (n = 213).

Considerations and location characteristics	n	%[a]
Calm; sheltered from wind	123	57.7
Shallow water	96	45.1
Many fish	53	24.9
Good water quality; clear, clean	52	24.4
Muddy sediment	42	19.7
Always has water; does not dry out	41	19.2
Water and/or mud is warm	36	16.9
Less frequented by fishers	33	15.5
Sandy sediment	29	13.6
Close proximity to marsh	27	12.7
Locations that can be easily monitored	22	10.3
No or little mud	14	6.6
Consult fishers affected	10	4.7
Other (slope of sediment; no water hyacinth)	7	3.3

[a] Proportions sum to >100% because 185 respondents (87%) provided multiple suggestions.

When fishers were asked for their view about the prospects for fishing at Lake Alaotra 5 years and 20 years into the future if restricted areas were well-enforced, all but one believed fishing would be better while 30% stated fishing would be much better after 20 years. Under this scenario, a significant number of fishers had changed their view compared to background interviews and stated that under these improved conditions they would want their son to become a fisher (McNemar's test $\chi^2 = 20.93$, $df = 1$, $p < 0.0001$). However, despite the perceived improvement in prospects for the fishery, most fishers (80%) still did not want their sons to become fishers. This sentiment may be tied to a traditional Malagasy adage "*Izay adala no toa an-drainy*" that means "*Foolish is he who seeks only to emulate his father*".

All except two fishers completing scenario interviews (n = 221) suggested a best timing and duration for a closed period. Eleven fishers suggested a completely different time of year to the current period, mainly between May to July, and for ten of these fishers the reason was to correspond with availability of work in rice fields. The remaining suggestions (n = 208) were grouped into three categories relative to the current start time of 15 November: i) Earlier: starts before 15 November (36%), ii) Same: starts on 15 November (46%), and iii) Later: starts after 15 November (18%). Reasons for these preferences differed between starting times (Fisher's exact test, $p < 0.0001$). A greater proportion of fishers who suggested an earlier starting time for a closed period stated this would be better for protecting spawning fish (93%; Figure 7.4). Conversely, a greater proportion of fishers suggesting the same or a later starting time preferred a closed period to coincide with availability of work in rice fields.

Although the majority of fishers (60%) believed a closed period should continue to span two months, 38% stated it should be only one month and 2% suggested it should be longer than two months. Most fishers (84%) suggesting the duration of a closed period should continue to be two months perceived this as the amount of time necessary to (a) cover the reproductive cycle of tilapia (i.e., lay eggs, eggs hatch, and adults protect fry) and (b) allow fry an opportunity to distribute as water level rises. Of 80 fishers stating that a closed period should be one month, 65% perceived this to be sufficient time to protect reproducing fish; the remaining 35% of these fishers believed a one-month closure would be easier for monitoring and for fishers to cope with financially.

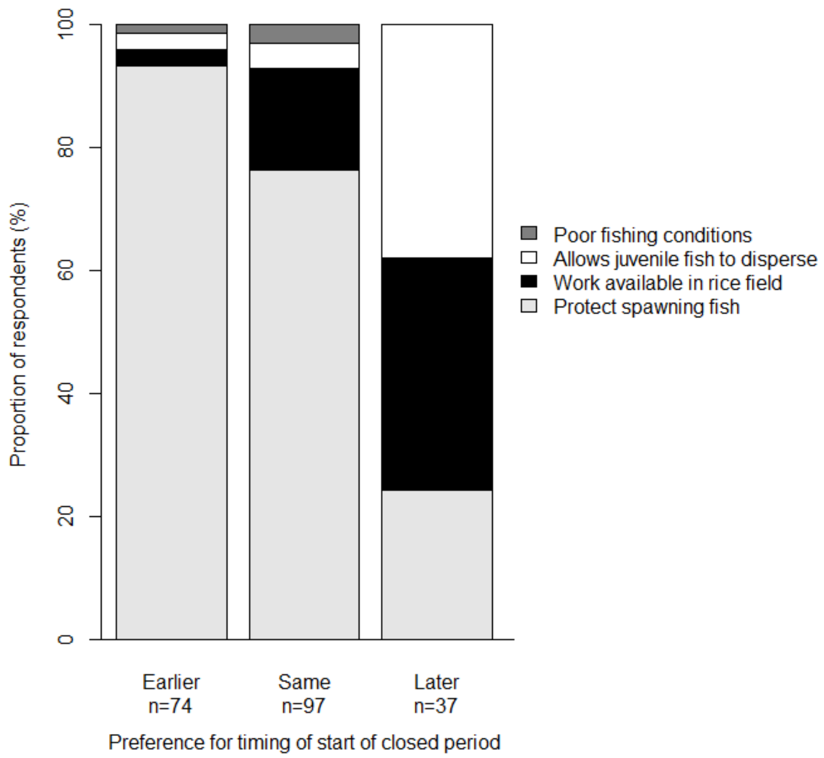


Figure 7.4. Fisher preferences for timing of start of closed period and reasons for preference. Fisher responses (n = 208) were grouped into three categories relative to the current start time of 15 November. 'Earlier' starts before 15 November, 'Same' starts on 15 November, and 'Later' starts after 15 November.

When fishers (n = 221) were asked to describe effects on the fishery if there were no regulations, 52% stated that the fishery would collapse and fishing would not be a viable livelihood; nine of these fishers perceived that without regulations fishers would use any method at their disposal to catch fish, which would destroy resources. Similarly, 47% of fishers stated that fish numbers would decline and others (1%) suggested there would be increased competition and fighting between fishers. Fishers were then asked how an absence of regulations would affect them; 95% stated they would lose their livelihood and income, have a lower standard of living, life would become very difficult, and/or they would have no future, while the remaining 5% of fishers expected reductions in catch size or increased conflict among fishers.

A common theme during discussions with fishers, particularly in focus groups, was that fishers perceived they could not govern the fishery themselves and would require assistance from authorities to implement and enforce regulations. Fishers stated that a complicating factor in transferring greater management authority to communities, as planned for Lake Alaotra, is that it creates social conflict which affects interactions and life in the village adversely. However, fishers confirmed that they would like to have fishing regulations enforced, and wished to be consulted for advice and involved in managing the fishery. Most fishers (83% of 221) stated a willingness to participate in collecting data about their fishing activity and catches (as conducted via catch interviews during the study), provided the information was used by authorities to improve management of the fishery. Some fishers also stated that collecting this data would help them track what is happening in the fishery. Of 38 fishers not wanting to collect data, most (23) stated they did not have time, eight stated they were illiterate or did not know how to write, and six stated they were too old or too tired to participate; one fisher intended to stop fishing.

7.4 DISCUSSION

My research examines fishers' perceptions of management interventions and their impacts, as well as fishers' anticipated responses to management scenarios, to gain insights that can be taken into account to reduce the costs of interventions for fishers and thereby increase their acceptability and probable compliance. This builds on the premise that effective management of fisheries relies on understanding fisher knowledge, perceptions, and behaviour (Branch et al., 2006). My results show that despite some variation in responses to scenarios, as well as differences in type of gear used and fishing effort, Anororo-based fishers are broadly homogeneous in their perceptions and understanding of interventions and also their responses to scenarios of management and change. The fishers have a generally good but incomplete knowledge of current fishery regulations and the reason(s) for the regulations; they are also primarily supportive of the regulations, understand that the regulations are intended to benefit fishers, and have positive expectations of the outcomes of future restrictions for their returns from fishing.

Few fishers correctly stated the locations of all restricted areas and this was the main subject of uncertainty in their knowledge of the fishery regulations. However, this is reasonable given that most Anororo-based fishers do not fish throughout the system but rather within a limited number of locations often determined by the type of gear used (see Chapter 6). Other areas of uncertainty or inaccuracy were the authorities responsible for managing the fishery and gear-specific restrictions in mesh size. It is possible that many fishers stated minimum legal mesh sizes smaller than specified in the fishery regulations in order to convey to the research team the impression that the (illegal) mesh size they use is legal. Although numerous fishers were observed to use under-sized mesh, they probably did not want to appear to be knowingly infringing the regulations, possibly because this may be perceived by other fishers, community members, or researchers as socially unacceptable (see Fisher, 1993; King and Bruner, 2000; cf. Milfont, 2009). Allison and Ellis (2001), Béné et al. (2009), and Evans et al. (2011) refer to similar stated uncertainty about regulations and governance within artisanal fisheries, which suggests it may be characteristic and a factor for managers (and researchers) to take into account, especially regarding enforcement.

Although compliance with management interventions is critical to their effectiveness, it is unlikely if resource users are unaware of regulations or do not understand them (Honneland, 1999; Keane et al., 2011; Peterson and Stead, 2011). Because Anororo-based fishers are generally aware of the fishery regulations and support them, most issues of compliance within the study area are probably tied to the impacts and costs to fishers of each specific intervention. For example, more than two-thirds of fishers who were aware of temporal restrictions continued to fish during the closed period in order to avoid economic hardship. Similarly, higher direct costs and fewer adaptive options probably explain why fishers viewed the closed period more negatively than spatial restrictions (also see Salas and Gaertner, 2004; Peterson and Stead, 2011). Analysis of fishers' perceptions and responses provides insights about the probable impacts and costs of interventions, making it possible to design them for greater compliance.

Fisheries are a combination of natural resource, human, and management subsystems that interact dynamically (Charles, 1995; Seijo et al., 1998). Fishers operate within the human subsystem, and respond and adapt to changes in other subsystems according to their capacity to adjust (Salas and Gaertner, 2004). While Anororo-based fishers were broadly similar in their perceptions and understanding of management interventions, variation in responses to scenarios provides further evidence that adaptive strategies and decisions may be constrained by individual objectives, feasible economic options, and/or socio-cultural factors (see Béné, 1996; Coulthard, 2008; Daw et al., 2011b). The adaptive responses expressed by Anororo-based fishers also confirm that they should not be regarded as fixed or inert elements of the Lake Alaotra fishery, but as central actors who are likely to influence system dynamics as much as they are impacted by those dynamics. This is consistent with the established but still poorly implemented finding that fishers' behaviour must be thoroughly understood and taken into account to manage fisheries effectively (Hilborn, 1985; Salas and Gaertner, 2004; Daw, 2008; Fulton et al., 2011).

This study provides support for the management assumption that fishers would shift fishing location if their current fishing location became a restricted area, rather than continuing to use that location and fishing contrary to regulations. Although the new locations nominated by fishers involved greater effort costs in travel time for some fishers but reduced effort costs for others, all new locations permitted continued use of the fisher's current type of gear. This suggests that the high and often prohibitive costs of changing fishing methods, in terms of capital investment as well as probable smaller catches due to less skill with different gear, are a primary factor in fishers' response decisions; this will be particularly the case for fishers who would also incur increased effort costs in response to interventions, either in time travelling to a new location or time to learn to use new gear (also see Gell and Roberts, 2003; Salas and Gaertner, 2004). Similarly, Daw et al. (2011b) note that fishers generally perceive that the economic and social costs of restricted areas outweigh their benefits. This was not explicitly the case for Anororo-based fishers, although it might be if fishers felt they were compelled to change gear as well as locations in response to spatial restrictions.

While high adaptive capacity is typically considered favourable for resource users in dynamic social-ecological systems (Adger and Kelly, 2001), many adaptive responses benefitting users over the short-term may compromise longer-term system resilience and sustainability (Cinner et al., 2011). Accordingly, Cinner et al. (2011) distinguish adaptive responses that increase or accelerate resource depletion (termed amplifying responses) from those that reduce resource depletion (termed dampening responses). Whereas Cinner and colleagues applied this distinction to system-level resilience and impacts on marine resources, in my study the terms amplifying and dampening refer to changes in the amount of effort that fishers would invest in fishing in response to interventions or management scenarios. This refinement in terminology facilitated a focus on fisher behaviour, and was adopted because not all examples of amplifying responses necessarily accelerate resource depletion; conversely, actions classified by Cinner et al. (2011) as dampening responses might increase rather than decrease resource depletion. For example, many Anororo-based fishers who stated they would adopt dampening actions (i.e., reduced effort or stop fishing) in response to impacts of greater fishing intensity also indicated that they would work in rice cultivation as an alternative or supplement to fishing; this was also the case for fishers seeking alternative work due to the perceived impacts of spatial or temporal restrictions. Although such actions are clearly dampening responses in terms of fishing pressure, the lack of available land for agriculture near Anororo would probably increase pressure to convert the marsh into rice fields, with increasingly adverse impacts for the fishery, as well as local biodiversity, over time. The net result of dampening responses by fishers would probably be amplification of overall resource depletion, using the categorisation by Cinner et al. (2011); in fact, the negative effects could be greater than if fishers had continued to fish as usual or increased effort. This is a key consideration, given that conversion for agriculture often drives wetland loss and is a major source of conflict between resource users and conservation managers (Rebelo et al., 2010; Finlayson et al., 2012). Similarly, fishers who would change their fishing location and/or method (an amplifying response in the categorisation by Cinner and colleagues) may have a reduced impact on the fishery if they fish away from spawning grounds or obtain a better return that allows them to fish less.

In line with findings reported by Cinner et al. (2011) for marine fishers in Tanzania, there appeared to be a ratcheting effect in the scenario responses of Anororo-based fishers. Most fishers anticipated amplifying responses requiring a greater investment in fishing, which are therefore more likely to be relatively permanent or long term, whereas dampening responses were anticipated less frequently and were often short term, reversible, or dependent on options. Both studies also found heterogeneity in responses between individuals, particularly according to type of gear used. The range of responses was consistent with other studies, including continuing to fish as usual despite reduced catches, moving to new locations, changing effort, changing gear or tactics, or seeking alternative sources of income, either temporarily or permanently (Allison and Ellis, 2001; McClanahan et al., 2005; Cinner et al., 2008).

Many management interventions discount fishers' responses and assume displaced fishing effort will be absorbed into the broader economy or disappear (Wilen et al., 2002; McClanahan, 2007). However, the predominance of amplifying responses in my study indicates that this is unlikely to occur. Displaced effort is more likely to be redistributed and intensified within the fishery, and this could be especially the case when there are few livelihood options or options are limited by poverty, education, or access to key resources such as land (see Béné et al., 2003; Cinner et al., 2008). The emphasis on amplifying responses under improving conditions (such as less fishing intensity) and also worsening conditions (such as smaller catches) is likely to provide challenges for sustainable management.

Amplifying responses to reductions in catches arising from interventions or changing conditions may have persistent detrimental impacts on fish stocks, further reducing catches and driving effort higher in a destructive cycle (Munro, 1983; Teh et al., 2007; Peterson and Stead, 2011). There is a range of management options to try to forestall such a cycle. Historically, a common option would be to introduce catch quotas and gear limits to improve fishery sustainability (see Munro, 1983); however, these measures are rarely suitable for artisanal fisheries and their efficacy would be equally contingent on fishers' capacity to adapt. Rather than enforcing reduced fishing effort or increased mesh size (Munro, 1983; Caddy, 1999), recent studies suggest that flexible spatial and temporal closures attuned to fishers' needs and behaviour could be more effective (Teh et al., 2007; Grantham et al., 2008; Game et al., 2009). Responses by Anororo-based fishers indicate that flexibility and tuning will be key to success. It is generally established that spatial restrictions on fishing can increase catches considerably in adjacent areas (Gell and Roberts, 2003; Cucherousset et al., 2007; Suski and Cooke, 2007); Roberts et al. (2001) found that a series of five small reserves in St Lucia increased artisanal fishers' catches in adjoining areas by 46% to

90% depending on gear used. However, benefits of spatial closures only accrue over time. In the short term it is essential to account for (a) displacement of fishing effort and associated costs for fishers as well as (b) potential escalation of effort due to smaller catches arising from greater fishing intensity outside restricted areas. The benefits of spatial closures over the long term will depend on how well these issues and the dynamics of change are addressed (see Kellner et al., 2007; Jennings, 2009; Valcic, 2009); compliance may be compromised if they are not resolved (Daw et al., 2011b).

As expected, and consistent with findings by Cinner et al. (2008) for fishers in nine coastal communities in Kenya, the proportion of fishers anticipating that they would stop fishing in response to greater fishing intensity increased with amount of decline in catch. However, the relationship was less pronounced than observed by Cinner and colleagues and there were threshold effects; Anororo-based fishers were most likely to exit the fishery under substantial (i.e., four-fold) increases in fishing intensity with 60% or greater reductions in catch. Dampening responses were a last resort for most fishers and would typically be preceded by amplifying actions; they also frequently depended on having other options for income, as noted by Ikiara and Odink (2000) and Cinner et al. (2011). Differences in fishery exit criteria for Anororo-based fishers compared to other studies confirm that socio-cultural factors may play a large role in fishing decisions, which are also site and context specific (Daw et al., 2012).

Intuitively, temporal restrictions could be viewed as more equitable between fishers than spatial restrictions because they limit all fishers irrespective of type of gear and effort used; in contrast, spatial restrictions are likely to impact some fishers (currently fishing in those locations) or gear types (for example, traps if restricted areas adjoin marsh) more than others, except to the extent that crowding and fishing intensity are increased by effort displacement. However, across gear types Anororo-based fishers perceived more negative aspects of the closed period than spatial restrictions. The key shortcoming of the closed period as currently scheduled was lack of income for two months; many fishers recommended bringing it forward to better protect spawning fish and thereby increase catches over the remainder of the year, or to postpone it to coincide with availability of alternative work in rice fields. It is also unrealistic and unreasonable to expect fishers to switch to using only the uncommon line & hook method during the closed period, particularly as this can only be used in marsh areas (which would then be overcrowded and also difficult to access in low water), targets only blotched snakehead with relatively low catch per unit effort, takes a heavy toll on reeds for fishing poles, relies on funds to purchase hooks and line, and requires fishers to collect frogs or worms as bait. The regulations for the closed period are therefore an example of an intervention poorly aligned with fishers' needs or adaptive capacity, which accounts for the low levels of compliance.

A key finding of this research is that almost all Anororo-based fishers perceived an absence of fishing regulations would deplete or collapse the fishery and have very negative livelihood impacts. Fishers were in favour of regulations and management interventions generally, although, as commonly reported in similar studies (see Béné et al., 2009; Eggert and Lokina, 2010; Peterson and Stead, 2011), fishers perceived their efficacy is undermined by low compliance, poor enforcement, or corruption by authorities. Experienced fishers also perceived that the existing regulations for Lake Alaotra's fishery are out of touch with prevailing conditions and fishers' needs (also see McClanahan et al., 2005).

Management interventions are more likely to be viewed favourably by fishers when designed and introduced in collaboration with them (Hilborn and Walters, 1992; Hart and Pitcher, 1998; Berkes et al., 2001; Salas and Gaertner, 2004), preferably within a mutually-legitimate co-management arrangement (Jentoft et al., 1998; Jentoft, 2000; Berkes, 2003). Co-management has been shown to benefit fisheries across all scales in terms of increased fish stocks and catches, improved livelihood returns, and greater compliance; it also gives resource users a voice and can finely-tune management to local contexts (Sverdrup-Jensen and Raakjær Nielsen, 1998; Gelcich et al., 2005; Cinner et al., 2012). Co-management also recognises that fishers are part of a broader community network, allowing traditional knowledge and forms of governance, such as taboos and customary norms, to be built into management practice (Berkes, 1985; Aswani and Hamilton, 2004a; Salas and Gaertner, 2004; Coulthard, 2008; Cinner et al., 2009a).

Anororo-based fishers are generally in favour of co-managing the fishery, believing the fishery's problems are too large for only either fishers or authorities to address. Fishers want to improve regulations to take costs to fishers into account in ensuring fishery sustainability; they also want to have regulations enforced. Fishers identified issues to be addressed by management, including repositioning no-take zones into a network of smaller more-effective areas less costly to fishers, rescheduling the closed period to better protect spawning fish, eliminating prohibited fishing gear, removing invasive aquatic plants that are degrading the marsh, and improving communication with management authorities. These statements demonstrate that artisanal fishers are often willing to be active partners in managing their fishery for sustainability rather than exploitation. Importantly though, fishers called on authorities to actively help in enforcing regulations because policing neighbours leads to social conflict within the village, indicating that this should be a core consideration for co-management plans.

Fishers' willingness to collect activity and catch data to monitor the fishery further demonstrates their keenness to co-manage. Although participatory monitoring is often an effective way to (a) engage resource users in interventions, sustainable practices, and conservation, (b) build legitimacy and relevance, (c) incorporate local knowledge and experience, and (d) construct larger data sets than otherwise feasible (Ticheler et al., 1998; Berkes et al., 2001; Obura, 2001; Pomeroy and Rivera-Guieb, 2005; Brenier et al., 2011), it should not be viewed primarily as a means of obtaining management information at low cost (see Danielsen et al., 2007; Danielsen et al., 2009).

As noted by Cinner et al. (2011), the hypothetical nature of scenario-based research means that observed behaviour, interactions with others, and changes in conditions must also be considered when making management decisions. However, the insights provided through scenarios would otherwise rarely be available at the planning stages of interventions. Advance knowledge of costs to fishers and their probable responses should allow interventions to be attuned to fishers' (and the fishery's) needs, thereby increasing the likelihood of the interventions being complied with and effective. My experience in this research is that scenario data are most likely to be sufficiently reliable if collected with (a) a comprehensive understanding of the fishery and fishing activity, and (b) the trust and confidence of fishers. Crucially, scenario approaches give fishers a voice in policy processes, which means their perceptions and interests are less likely to be ignored or overlooked when change is required (Salas and Gaertner, 2004). This study is one of few exploring such issues within an artisanal fishery in a developing country (Cinner et al., 2008); it also affirms that understanding fisher behaviour and costs to fishers is key to management efficacy (Wilen et al., 2002; Hilborn, 2007; Daw et al., 2011b). Although the current outlook for Lake Alaotra's fishery is challenging from a management perspective, it is encouraging that fishers are largely aware of problems to be addressed, generally in favour of regulations and enforcement, and willing to contribute to solutions if their costs and concerns are taken into account.

SUPPORTING INFORMATION

Supporting information for this chapter is in Appendix S7 and includes fisher adaptive responses to: i) a change in fishing intensity grouped by age category (S7.1), ii) a change in the frequency of good and bad fishing days (S7.2), and iii) the removal of invasive aquatic plants (S7.3).

CHAPTER 8 INCLUDING FISHERS IN CONSERVATION PLANNING: THE COSTS OF ALTERNATIVE RESERVE NETWORKS

8.1 INTRODUCTION

Protected areas or no-take zones can increase fishing yields in adjacent harvested areas over the long-term but also generate short-term costs for displaced fishers (Roberts et al., 2001; Gell and Roberts, 2003; Hilborn et al., 2004; Béné et al., 2007; Cucherousset et al., 2007; Suski and Cooke, 2007). Selection of protected areas is typically based on key ecological criteria, while factors such as the economic value and social importance of the resource for local people are often ignored (Smith and Wilen, 2003). These oversights are likely to compromise the effectiveness of the protected areas over time (Beger et al., 2004; Johannesen, 2007), primarily because conservation strategies are more likely to be accepted and implemented successfully when the courses of action involved are transparent to all stakeholders and designed with the needs of local people in mind (Sutherland, 2000; Pierce et al., 2005; Knight et al., 2006b; Margules and Sarkar, 2007). A comprehensive understanding of the socioeconomic system driving resource use is essential for effective management; if incorrect assumptions are made, an inappropriate and ultimately unsuccessful strategy may be selected (Muller and Albers, 2004; Johannesen, 2006). Recent research emphasizes the critical importance of incorporating spatially-explicit socioeconomic information into conservation planning to provide more accurate assessments of the potential impacts of different management scenarios and avoid costly, and possibly irreversible, mistakes (Bode et al., 2008b; Carwardine et al., 2008; Polasky, 2008; Ban et al., 2009; Game et al., 2011).

Although fisheries managers are increasingly concerned about the livelihoods of people who rely on fishery resources (Berkes, 2003; Welcomme, 2003), consideration of the costs of conservation planning has typically focused on the acquisition, implementation, and ongoing management costs of protected areas, while costs to resource users have rarely been quantified in meaningful ways (see Frazee et al., 2003; Naidoo et al., 2006; Carwardine et al., 2010). Frazee et al. (2003) suggest that unrealistic estimates of the costs of implementation and ongoing management of conservation interventions (including opportunity costs to local people) could lead to underfunded ‘paper parks’ that fail to meet conservation goals; other researchers argue that focusing too strongly on the economic costs of conservation planning will lead to sub-optimal reserve selection (see Arponen et al., 2010). Socioeconomic costs included in conservation planning to date have focused primarily on opportunity costs such as foregone benefits from alternative uses of land or time (Stewart et al., 2003; Cameron et al., 2008; Ban and Klein, 2009; Adams et al., 2010). However, Adams et al. (2010) found that opportunity costs can inadvertently shift conservation costs among groups of stakeholders. While there is no argument that opportunity costs are important, these must be balanced with direct costs for resource users, which may need to take precedence in more market-driven areas. Choosing an appropriate approach will depend on the context of the planning problem (Cameron et al., 2008).

The Lake Alaotra wetland is Madagascar’s primary inland fishery and also a site of considerable biodiversity conservation importance, primarily because it provides the only habitat for the critically endangered Alaotran gentle lemur (*Hapalemur alaotrensis*). A suite of fishing regulations (such as an annual two-month fishery closure, mesh size restrictions, and minimum size limits for fish caught) have been implemented (Dina de Pêche, 2006) but with little compliance. The current Lake Alaotra Management Plan, developed by the Durrell Wildlife Conservation Trust (DWCT) in conjunction with the regional Fisheries Service in 2006, fails to consider the costs of compliance for resource users. Attempts to implement and enforce the management plan have not been successful, and all stakeholders (i.e., DWCT, the Fisheries Service, and local fishers) now consider the plan inadequate to address biodiversity conservation, fishery management, and social objectives. Because temporal closures are frequently less biologically effective than spatial closures (Grantham et al., 2008), and also cause socioeconomic hardship for fishers, it has been proposed to phase out Lake Alaotra’s closed period in favour of spatial restrictions (R. Lewis, DWCT, pers. comm.). Consequently, improved spatial planning that incorporates a social dimension is crucial.

I used systematic conservation planning software to combine ecological data with information on catch weight and fisher behaviour to develop potential reserve networks that minimise costs to fishers subject to meeting biodiversity targets. Three biodiversity layers were used in the analysis, which in total broadly represented the management goals for the region (see Ranaivonasy et al., 2005): 1) protecting key habitat for the Alaotran gentle lemur; 2) protecting higher quality marsh vegetation for overall biodiversity; and 3) improving fishery sustainability by establishing no-take zones that protect nursery areas for spawning and juvenile fish. Marxan (Ball and Possingham, 2000) and the MinPatch extension (Smith et al., 2010) were used to find optimal reserve solutions that minimised costs to fishers, while ensuring targets to protect lemur habitat, high quality vegetation, and fish nursery areas were achieved. I used two forms of cost to fishers, which represented the relative economic values of planning units based on catch weight (termed ‘catch cost’) and the relative costs of accessing planning units in terms of distance from the village (termed ‘proximity cost’). The resultant reserve configurations were then compared with the proposed management plan in terms of area reserved and cost to fishers. Priority areas for conservation were identified and the sensitivity of the outcome to the two forms of cost were examined by adjusting the relative importance and trade-offs between them.

8.2 METHODS

8.2.1 The planning area

Lake Alaotra is the largest lake in Madagascar and base for the nation’s most productive inland fishery (Andrianandrasana et al., 2005). The lake covers an area of 200km^2 (40km long and width varying from 3km to 8km), marsh adjacent to the lake covers an area of 230km^2, and approximately 1,200km^2 of rice fields adjoin the lake and marsh (Andrianandrasana et al., 2005; Ferry et al., 2009). Detailed information about the study site and fishery characteristics is provided in Section 3.1 in Chapter 3 as well as Chapter 4.

Catch data used in this study were collected in June and July 2009 and for 13 months from October 2009 to November 2010 in Anororo village, which is a relatively large community of approximately 8,000 people on the west side of Lake Alaotra (PCD, 2004). Anororo village is adjacent to marsh and lake habitat and was selected for its (i) large population of fishers using a variety of habitats and fishing methods, (ii) proximity to planned conservation interventions, and (iii) local dependence on fishery resources for subsistence, income, and commercial activity. Fine-scale catch data were collected using structured catch interviews with Anororo-based fishers returning from their fishing trip. Semi-structured interviews, focus groups, and fisher follows were also conducted with fishers to gain further knowledge and understanding of the drivers of fisher behaviour, fisher perceptions of management interventions, and the perceived impacts and/or benefits of interventions for fishers (see Chapters 6 and 7). This additional contextual information aided the parameterisation of Marxan and MinPatch. Respondent codes were assigned to all participating fishers to preserve their anonymity (Bernard, 2002).

Because catch data were collected only from Anororo-based fishers, it was valid to define the planning area as that covered by the fishing locations used by these fishers and represented within the dataset. The lake and marsh were clipped 1km north, east, south, and west from the furthest fishing location in each direction from the village and for which catch data were collected ($n_{locations}$ = 55). Twelve additional geo-referenced fishing locations used by Anororo-based fishers are encompassed within the planning area; however, catch data had not been collected from these locations. The planning area is therefore representative of the fishing area used by Anororo-based fishers. Some fishers from other villages also fish in parts of the planning area on a limited basis, typically relatively far from Anororo. Other stakeholders such as marsh-product users harvest marsh plants within the planning area. However, this study focuses on costs to Anororo fishers only, and a separate exercise (including fine-scale data collection) would be required to account for costs to (a) fishers from other villages or (b) other stakeholders. Although fishers from all villages around the lake fish in similar ways and probably have similar catches, proximity costs would vary between villages.

The planning area was divided into 26,840 planning units to form 100m x 100m (1ha) grid cells. Due to the irregular shape of the lake and surrounding marsh, a number of planning units were truncated at the boundary resulting in some variation in planning unit size (area) and boundary length. Planning units less than the area of a Landsat pixel (30m x 30m) were deleted. In addition, planning units within lake habitat for which no targets were set were removed from the analyses, resulting in a total of 19,557 planning units (17,751ha) for the planning area.

8.2.2 Data layers

8.2.2.1 Biodiversity features and targets

8.2.2.1.1 The Alaotran gentle lemur

The Alaotran gentle lemur is endemic to the Lake Alaotra wetland and confined to marshes adjoining the lake. The lemur is therefore of high conservation interest both globally and locally, and has been afforded considerable research and conservation effort (see Mutschler et al., 1994; Mutschler and Feistner, 1995; Mutschler et al., 1998; Mutschler et al., 2001; Mutschler, 2002; Waeber and Hemelrijk, 2003; Andrianandrasana et al., 2005; Ralainasolo et al., 2006; Guillera-Arroita et al., 2010b; Lahoz-Monfort et al., 2010; Hudson, 2011). Estimated population size for the lemur decreased from more than 10,000 individuals in 1994 (Mutschler and Feistner, 1995) to less than 3,000 in 2002 (Ralainasolo, 2004); however, the species is highly cryptic, making it difficult to accurately assess population status, and recent research suggests the population may exceed the 2002 estimate (Guillera-Arroita et al., 2010a).

Lahoz-Monfort et al. (2010) developed a habitat suitability map for the Alaotran gentle lemur, using a combination of maximum entropy habitat suitability modelling and Landsat7 satellite imagery (Figure 8.1). Their study generated habitat suitability estimates for each 30m x 30m grid cell within the southern Alaotra marsh and I used those estimates that were within the planning area when setting conservation targets.

To avoid the loss of information on habitat suitability and potential distribution that can arise from setting arbitrary thresholds (Polasky et al., 2000; Wilson et al., 2005), I used the continuous index of habitat suitability generated by Lahoz-Monfort et al. (2010), applying two conservation targets (high: 50% and low: 30%) expressed as a percentage of total habitat suitability and based on estimated lemur population and average territory sizes. Average lemur group size is four individuals (Nievergelt et al.,

2002) and they occupy home ranges of 0.6ha to 8ha (Mutschler and Tan, 2003) with an average territory of approximately 2ha (Mutschler et al., 1994). The high target of protecting 50% of all habitat suitability equates to approximately 2,535ha, which would protect an area sufficient for approximately 5,000 individual lemurs. Given the lemurs' persistence to date, this may be a basis for a potentially viable population. To be conservative, the minimum size of a reserve patch in the marsh zone for lemurs was set to 8ha (i.e., 8 planning units), corresponding to their maximum home range.

8.2.2.1.2 Marsh vegetation

The Alaotra wetland is internationally recognised as an important area for biodiversity conservation and was declared a Ramsar site in September 2003 (Ramanampamonjy et al., 2003). The wetland provides habitat for 30 species of waterfowl including the endangered Madagascar pochard (*Aythya innotata*) and Meller's duck (*Anas melleri*); it also supports 15 species of fish, including rare and endemic species such as the Madagascar rainbowfish (*Rheocles alaotrensis*), tilapia sp. (*Paratilapia polleni*), and freshwater goby (*Gobius aenofuscus*) (Ramanampamonjy et al., 2003). A new species of small carnivore, Durrell's vontsira (*Salanoia durrelli*), is also known to inhabit the wetland (Durbin et al., 2010). Accordingly, because the wetland's marsh habitat provides food, shelter, and breeding sites for many species, it is a key biodiversity conservation component in the planning area.

Based on 2007 Landsat imagery, Andrianandrasana (2009) developed a vegetation classification map for the marshes of Lake Alaotra (Figure 8.2). Ten distinct habitat types were identified and defined (Table 8.1), and each 30m x 30m grid cell was classified into one of these types. The sum of the amount of each vegetation type within each planning unit was then used when setting conservation targets. Although this biodiversity feature was derived from the same Landsat imagery used for the lemur feature, the two can be considered independent because the latter feature was also derived from additional modelling.

There is considerable debate about what proportion of an area should be protected to achieve conservation goals. For landscapes, a target of 10% for each vegetation type is frequently used but also widely criticised (McNeely, 1993; Soulé and Sanjayan, 1998). The IUCN World Parks Congress recently suggested that a 30% target would be more appropriate (IUCN, 2003); however, Soulé and Sanjayan (1998) suggest that the minimum proportion of land area necessary to fully protect biodiversity is 50%. Nel et al. (2011) identified four key principles to consider when planning for the persistence of freshwater biodiversity. These are: 1) select areas of high ecological integrity, 2) incorporate connectivity, 3) incorporate areas important to population persistence, and 4) incorporate any additional natural processes that can be mapped. In order to select areas with intact high quality vegetation as well as areas important to the persistence of biodiversity, targets were aimed at protecting primarily high and medium quality papyrus. Vegetation targets are expressed as a percentage of the extent of each vegetation type. To explore the sensitivity of Marxan results to these targets, a second set of lower targets were also applied. While low quality papyrus is not highly desirable habitat in terms of protecting freshwater biodiversity, protecting a small proportion of this vegetation type in each scenario may allow for possible restoration or natural regeneration of the area as well as connectivity between areas.

Table 8.1. Classification key for vegetation and habitat types (Andrianandrasana, 2009).

Name (grid code)	Definition
Low quality reeds (4)	Dominated by papyrus, low density with only 1 or 2 species of flora, and height <2m
Medium quality reeds (5)	Dominated by papyrus, medium density with 3 to 4 species of flora, and height between 2m and 3.5m
High quality reeds (3)	Dominated by papyrus (≥80%; *Cyperus* spp.), robust, strong, high density, rich in floristic composition (at least 5 species present), with an average height of 3.5m
Reeds / rushes (7)	*Phragmites communis*
Giant arum (10)	*Typhonodorum lindleyanum*
Invasive plant A (1)	*Azolla* spp. or *Salvinia* spp.
Invasive plant B (2)	Water hyacinth *Eichhornia crassipes*
Savanna (8)	Grasses
Bog (9)	Bog
Water (6)	Open water; lake
No classification (0)	Classification could not be determined

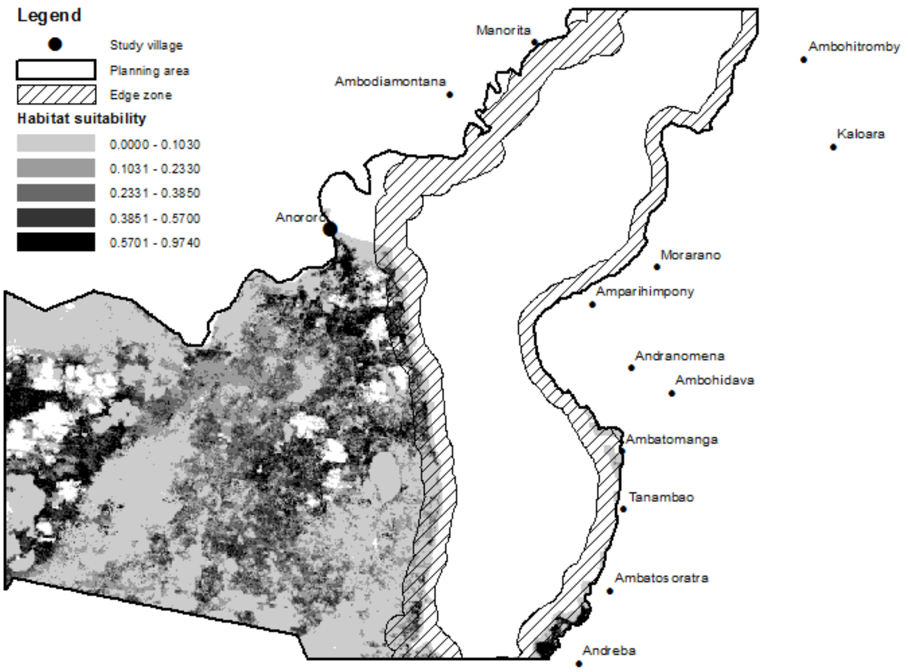


Figure 8.1. Habitat suitability map within the planning area for this study, developed by Lahoz-Monfort et al. (2010), showing suitability estimates from low (light grey) to high (black) suitability. Diagonal lines represent the edge zone.

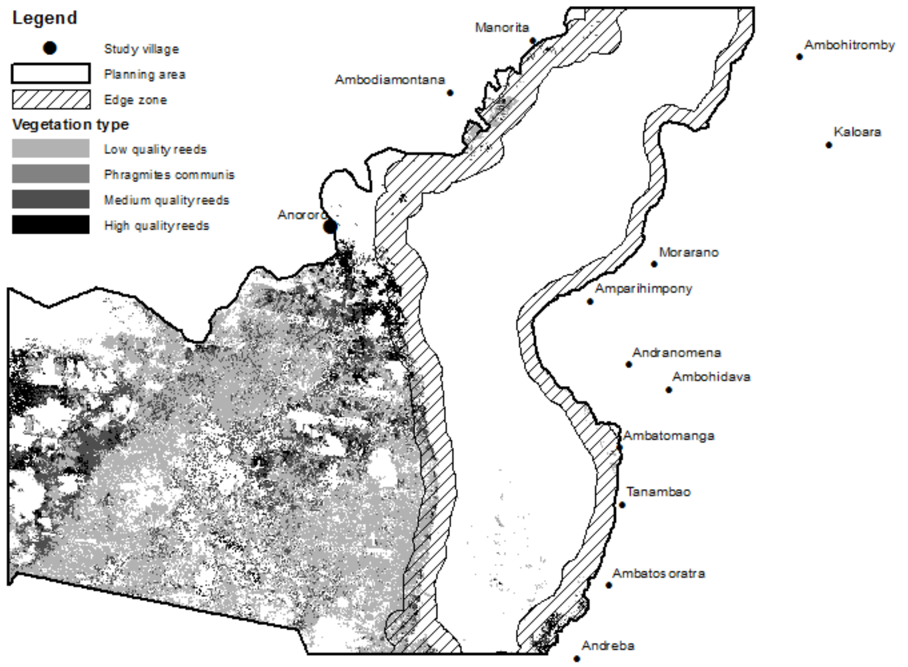


Figure 8.2. Vegetation classification map within the planning area, developed by Andrianandrasana (2009), showing vegetation types for which biodiversity targets were set in this study from lower (light grey) to higher (black) target vegetation types. Diagonal lines represent the edge zone.

8.2.2.1.3 Nursery areas / no-take zones (NTZs)

Littoral (edge) zones are known to be important nursery areas for fish because they provide access to food and shelter, have higher densities of plankton, and waves and turbulence regularly re-oxygenate the water (Welcomme, 1985). Protecting spawning and nursery areas was also the broad aim of the no-take zones proposed along the lake-marsh interface under the current management plan (Razanadrakoto and Rafaliarison, 2005).

Direct observation of fish eggs, fry, and juvenile Nile tilapia (*Oreochromis niloticus niloticus*; <13 cm in length) during catch interviews with fishers confirmed that the interface between marsh and open water is the most suitable nursery habitat for fish in Lake Alaotra. The majority (86%) of Nile tilapia <13cm in length caught during the study, and the majority (74%) of observations of eggs and/or fry, were from fishing locations within edge habitat. Fishers defined edge habitat in the planning area to be 500m into the lake from the lake-marsh interface and also, where marsh existed, 500m into the marsh (illustrated in Figures 8.1 and 8.2), and this definition was used throughout the study. This 1km-wide buffer at the lake-marsh edge is referred to as the nursery layer or edge zone.

Following the proposed Alaotra management plan (Razanadrakoto and Rafaliarison, 2005) in which 15% of the surface area of the lake was designated as a no-take zone (NTZ), the equivalent target within the planning area for this study totals 1,704ha (40% of the edge zone); a second lower target of 1,281ha (11% of the surface area of the lake or 30% of the edge zone) was also applied.

Two objectives for NTZs in the edge zone were included, based on the literature for suitable designs of no-take zones and after consultation with local fishers: i) have many smaller reserves rather than the few large reserves that exist in the current management plan; and ii) have reserves that are distributed relatively evenly around the lake edge. Despite a common perception that larger reserves are more effective in meeting biological goals, Halpern (2003) found that the relative impacts of marine reserves are independent of reserve size. A network of small, more-efficient reserves that are socially attainable and supported by local communities, and therefore have high levels of compliance, is likely to be more biologically effective than a system of large reserves that exist only on paper (Aswani and Hamilton, 2004b; Mills et al., 2010; Weeks et al., 2010a). Numerous small no-take zones placed evenly around the edge zone may also distribute the associated costs and benefits more equitably between resource users (Cinner, 2007).

I set the minimum NTZ size for fish nurseries in the edge zone at 4ha, which was based on fishers' responses during scenario interviews and the need to ensure that NTZs were of a biologically-relevant size (see Chapter 7). I found that fishers used a fishing area spanning a mean distance of 0.81km (range 0.2 to 2.5km) on a single fishing trip. Fishers' territory size was therefore approximated at 1km, which would be sufficient space for fishing between NTZs and is also easily navigable by fishers (i.e., it takes a fisher about 15 minutes to travel 1km in a canoe). Accordingly, I used 5km^2 grid cells to define blocks (n = 34) in the edge zone and specified that a minimum of 4ha (i.e., 4 planning units) had to be selected within each block to ensure NTZs were evenly distributed with an adequate minimum distance between them across the edge zone.

8.2.2.2 Cost layer

A static approach was used to develop a cost layer based on current fishing levels. I did not incorporate any measure of potential benefits from establishing no-take zones that might enhance fish abundance or catch because these are temporally distant and perceived immediate costs to fishers will have the greatest influence on the social acceptability of management interventions.

The cost layer combines two forms of cost: average catch weight per trip and proximity to village. Fishing locations with larger average catch weights per trip are more valuable to fishers in monetary terms and hence more costly if selected as part of a reserve network. Ninety-one percent of fishers interviewed stated they would change fishing location in response to spatial interventions (see Section 7.3.2.1 in Chapter 7), which validated a location-based focus. Catch interview data for all gear types were used in a generalised additive model (GAM) to predict the distribution of catch weight across the planning area (Table 8.2). GAMs with spatial references (longitude and latitude) are commonly used in fishery spatial modelling because of their flexibility in accounting for the subtleties and nonlinearities of the relationship between catch and the explanatory variables (Borchers et al., 1997; Venables and Dichmont, 2004b; Wood, 2006, 2008; Rupp et al., 2012).

Variables included in the final model were habitat, time spent fishing, and the longitude and latitude of the fishing location where the catch was made. Time spent fishing and the spatial covariates were significantly non-linear (see Figures S8.1 and S8.2 in Appendix S8). Although initial models also included distance from the village as an explanatory variable, this led to a 0.4% decrease in the variation explained, indicating that distance did not improve the model over and above the spatial patterns captured by the longitude and latitude coordinates. Since the aim was to reach a final best model to predict catch across the planning area with as few variables as possible, distance from the village was removed as a variable. The variation explained by a model that did not include longitude and latitude was only 13.7%, and increased to 20.8% when the spatial reference was added. Although 20.8% is not high, it is an expected level of variation explained by models for ecological datasets (Wood, 2006); the model is congruent to the linear mixed effects model developed in Chapter 5 and provides an intuitive value for catch across the planning area. The final model with the three explanatory variables can be expressed in the following form:

$$C = H + f(E) + f(long, lat)$$

where C is the predicted catch weight in grams per trip, H is the dominant habitat type for each planning unit cell, E is the effort used measured as time spent fishing on a given trip, and *long*, *lat* is the interaction between the longitude and latitude of a given location. The f term represents a smooth function, which in the case of effort is a cubic spline and in the case of the longitude-latitude interaction is a tensor product. A second GAM with time spent fishing as a function of longitude and latitude indicated that there was little or no variation in time spent fishing across the planning area and therefore a constant effort value was used for predicting catch weight in the final model.

Catch weight is readily converted into money by using the median price per gram of catch (1.8Ar/g; see Chapter 4) to derive catch cost (Figure 8.3a). The catch cost (C_{cost}) function is therefore written as:

$$C_{\text{cost}} = pC$$

where p is the price of fish per gram in Malagasy Ariary (Ar) and C is the catch weight in grams predicted by the GAM.

Table 8.2. Results of the generalised additive model to predict the distribution of catch weight across the planning area. [a] Baseline level is 'edge' habitat. Significant values are in bold.

Explanatory variables	Estimate	SE	t value	*P*
Parametric coefficients:				
Intercept	7.1650	0.0922	77.684	**<0.0001**
Habitat[a]				
Lake	-0.4002	0.6178	-0.648	0.517
Marsh	0.3651	0.2028	1.800	0.072
Smoothed terms:	**edf**	**Ref.df**	**F**	***P***
Time spent fishing	5.332	6.479	14.14	**<0.0001**
Latitude Longitude	13.521	14.955	10.18	**<0.0001**

R-sq. (adj) = 0.198 Deviance explained = 20.8%
GCV score = 1.1894 Scale est. = 1.1746 n = 1755

Proximity cost is the cost to the fisher of locating reserves close to the village, and was based on two assumptions. First, because fishers fished at higher density near the village, reserves located near the village would displace more fishers and therefore incur greater costs to the fishing community; 53% (n = 930) of fishing trips were within 4km of the village, representing 52% (n = 269) of fishers who participated in catch interviews, and the mean distance travelled across all fishers was 3.85km. Second, fishers often had other work to do and stated that they fished how and where they did to facilitate being able to do this additional work. Locating reserves near the village would force fishers to travel further, reducing their ability to carry out other activities. We captured proximity cost (Figure 8.3b) using a 3-parameter sigmoidal model (see Heinz et al., 2005), with proximity cost (P_{cost}) as a decreasing function of distance d from the village:

$$P_{cost} = a * (1 - \exp^{(-b*\exp^{(-d/c)})})$$

where a is a coefficient that achieves equivalent mean values for catch cost and proximity cost distributions (a was set at 10,000), b is easy daily paddling range, and c is mean trip distance. Easy daily paddling range, b, and mean trip distance, c, were both set at 4km (equivalent to one hour of travelling time), which represents the maximum distance a fisher would travel for a 'quick' fishing day.

Total cost combines the two types of cost (catch cost and proximity cost) into a single value (see Figure 8.3c). However, catch weight, and therefore catch cost, generally increases with increasing distance from the village, and conversely the greater the distance from the village the lower the proximity cost. Following Stewart and Possingham (2005), a weighted variable alpha (α) was included to explore trade-offs between catch cost and proximity cost, and adjust the relative importance of the two costs. Setting alpha (α) = 0 would include only proximity cost whereas α = 1 would place entire importance on catch cost.

Not all planning units are full size (1ha); planning units along the edge of the marsh or lake followed the natural boundary and were typically smaller. The total cost of an individual planning unit is therefore a function of its area, A. Total cost (T_{cost}) as a function of area, catch cost, and proximity cost is written as:

$$T_{cost} = A[\alpha C_{cost} + (1 - \alpha) P_{cost}]$$

where A is the area of the planning unit (ha), α is the cost weighting, C_{cost} is the catch cost, and P_{cost} is the proximity cost.

Six total cost layers were generated to explore the sensitivity of Marxan results to the weighting (alpha) of the two cost layers in calculating total cost; alpha was varied between 0 (all weight on proximity cost) and 1 (all weight on catch cost) at intervals of 0.2 (Table 8.3).

Table 8.3. Total cost layers with varying alpha for scenarios run in Marxan.

Total cost layer	**Alpha (α) value**	**Definition**
Tc0	0.0	All weight on proximity cost
Tc2	0.2	20% weight on catch cost; 80% on proximity cost
Tc4	0.4	40% weight on catch cost; 60% on proximity cost
Tc6	0.6	60% weight on catch cost; 40% on proximity cost
Tc8	0.8	80% weight on catch cost; 20% on proximity cost
Tc10	1.0	All weight on catch cost

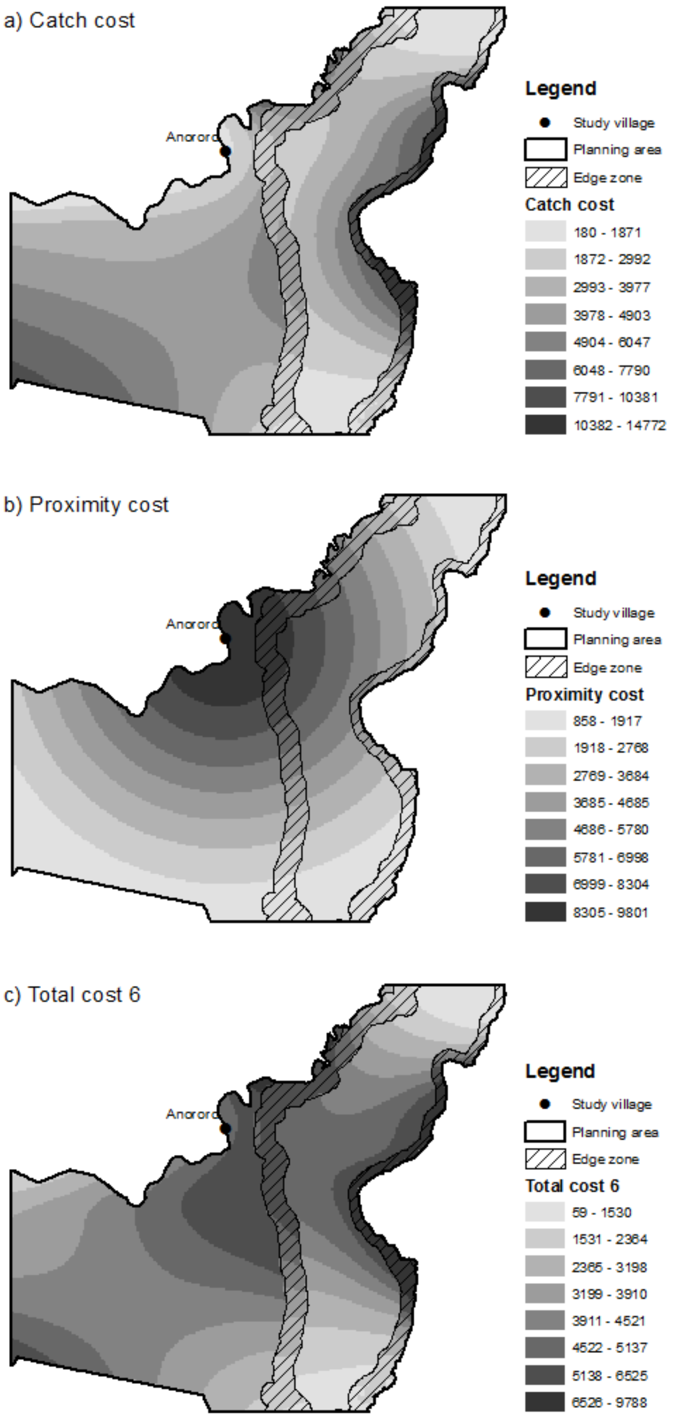


Figure 8.3. Maps showing the two forms of cost: a) catch cost and b) proximity cost, as well as an example of c) total cost (with alpha set at 0.6). Variations in costs across the planning area are from low (light grey) to high (dark grey) costs.

8.2.3 Running the planning scenario assessments

Marxan (Ball and Possingham, 2000) is advanced spatial analysis software used to support reserve design and planning of conservation areas worldwide. MinPatch (Smith et al., 2010) is extension software that uses Marxan results to find spatially-efficient solutions. Users, such as conservation managers, protected-area authorities, and government agencies, can employ Marxan and MinPatch to identify an efficient and effective reserve network that meets specified biodiversity and social targets at minimal cost.

I used the CLUZ (Conservation Land-Use Zoning) Arcview extension (Smith, 2004) to extract the amount of each conservation feature within each planning unit and to run scenarios in Marxan. The boundary length modifier (BLM) variable in Marxan is used to specify the degree of emphasis on minimising the overall reserve system boundary length; the higher the BLM value the more the boundary length will be minimised to produce a more compact reserve design (Game and Grantham, 2008). Based on a preliminary sensitivity analysis, I used a BLM value of 10 because this represented an acceptable trade-off between efficiency and cost (Stewart and Possingham, 2005).

Separate scenarios for each of the six total cost layers were run in Marxan with the sets of high and low biodiversity targets, resulting in a total of 12 scenarios (Table 8.4). Each scenario had 500 repeat runs with 2,000,000 iterations. From interviews with fishers I estimated that catch cost was more important than proximity cost from their perspective. Therefore, I used scenarios where alpha=0.6 (giving catch cost an importance weighting 1.5 times greater than proximity cost) for further analyses and comparison. However, the actual trade-off between the relative values of catch cost and proximity cost should be elicited from fishers more systematically for future planning and implementation.

Table 8.4. Targets for scenarios run in Marxan to develop a reserve network for the Lake Alaotra wetland. High, medium, and low quality vegetation refer to stands of reeds dominated by papyrus and Phrag. is the reed *Phragmites communis* (see Table 8.1). Six total cost layers (see Table 8.3) were run with each set of biodiversity targets for a total of 12 scenarios. The species penalty factor was set to 1,000,000 for all biodiversity features to ensure that Marxan met all biodiversity targets.

	Nurseries/ NTZs (%)	**Vegetation (%)**				**Lemur (%)**
Target	Edge zone	High qual.	Med qual.	Low qual.	Phrag.	Habitat suitability
High	40	70	50	10	30	50
Low	30	50	30	5	20	30

Marxan results were subsequently run in MinPatch. This additional step ensures the spatial configuration of reserves is clumped rather than scattered across the planning area. Using MinPatch is more efficient than using Marxan and the BLM value alone, particularly when seeking to produce reserve designs containing protected areas of a minimum size (Smith et al., 2010).

The Kappa statistic (Cohen, 1960) was used to assess the overlap between reserve designs and compare the results of different Marxan-MinPatch analyses. Cohen’s Kappa is a more robust measure of overlap than direct percentage agreement because it takes into account the amount of agreement which would be expected by chance. The Kappa statistic ranges from -1 to +1; negative values indicate poor agreement or disagreement, 0 indicates agreement due to chance, values from 0.01 to 0.2 indicate slight agreement, 0.21 to 0.4 fair, 0.41 to 0.6 moderate, 0.61 to 0.8 substantial, 0.81 to 0.99 almost perfect, and a value of 1 indicates perfect agreement (Landis and Koch, 1977).

8.3 RESULTS

8.3.1 Updating Marxan results with MinPatch

To ensure each reserve was above the minimum target size, Marxan results were run in MinPatch where minimum reserve sizes for the two protected area zones (marsh and edge) could be set as described above (Section 8.2.2.1). MinPatch reduced the number of reserve patches by an order of magnitude in all scenarios (Figure 8.4).

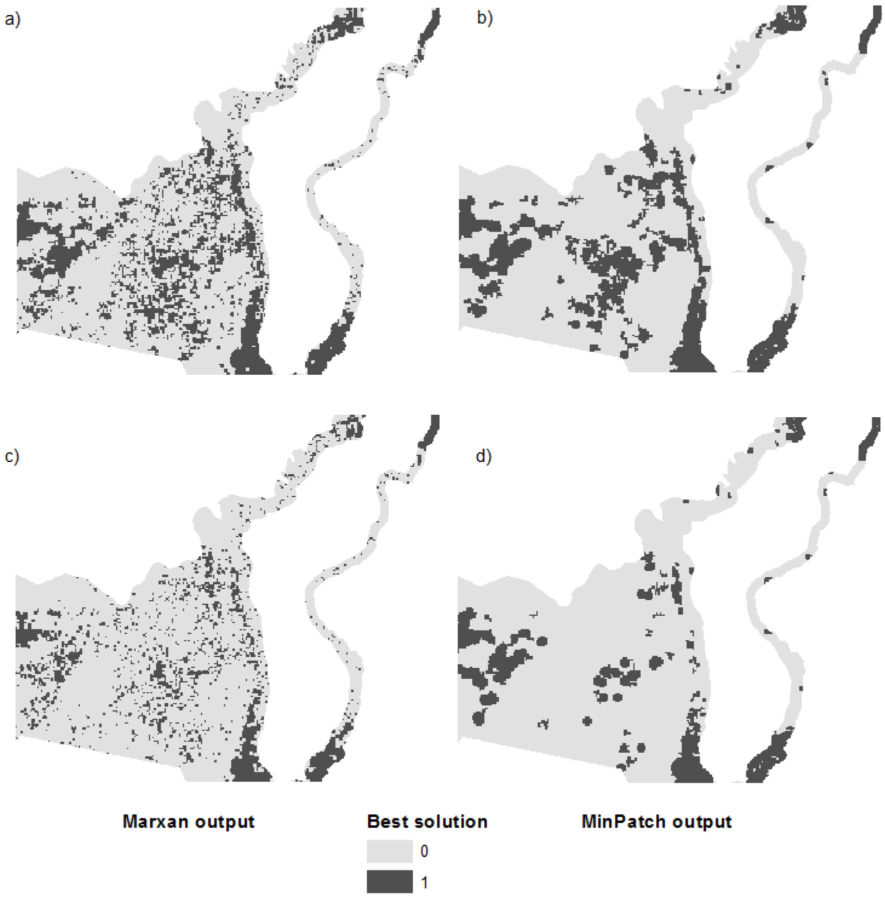


Figure 8.4. Best solutions produced by Marxan, based on two biodiversity target levels (high and low) and alpha = 0.6, and the results of updating them using MinPatch. Areas not selected in the reserve design are shown in light grey (0); areas selected for reserves are shown in dark grey (1). a) Marxan results high targets, b) MinPatch results high targets, c) Marxan results low targets, and d) MinPatch results low targets.

8.3.2 Sensitivity to cost

The sensitivity analyses indicate that outcomes varied depending on the weighting given to different components of cost. Under both high and low biodiversity targets reserve designs moved progressively northward and closer to the village as the weighting for proximity cost was decreased and the weighting for catch cost was correspondingly increased (Figure 8.5). Despite this pattern, planning units selected within the best reserve network solutions using different total cost layers overlapped substantially (Table 8.5), suggesting that the results were not overly sensitive to the cost weightings used. As expected, the greatest difference between reserve systems was at the extremes where only proximity (alpha=0) or catch cost (alpha=1) were included. Moderate to substantial agreement between the scenarios where alpha=0.6 and other weightings (Kappa statistics between 0.55 and 0.74) substantiates that the 0.6 weighting provides a suitable balance between the two forms of cost.

The overlap between the reserve systems under the different cost weightings indicates that they are not driven solely by cost but also by the underlying distribution of biodiversity features. I explore the influence of biodiversity features on reserve design further in Section 8.3.3. Furthermore, for a given set of targets there was minimal variation between scenarios in total area and cost of the alternative reserve systems (Figure 8.6). Although cost layers where alpha=0.6 and alpha=0.8 have the highest overall cost, they also have the smallest area, which indicates that the area of the reserve system decreases when greater weighting is assigned to catch cost.

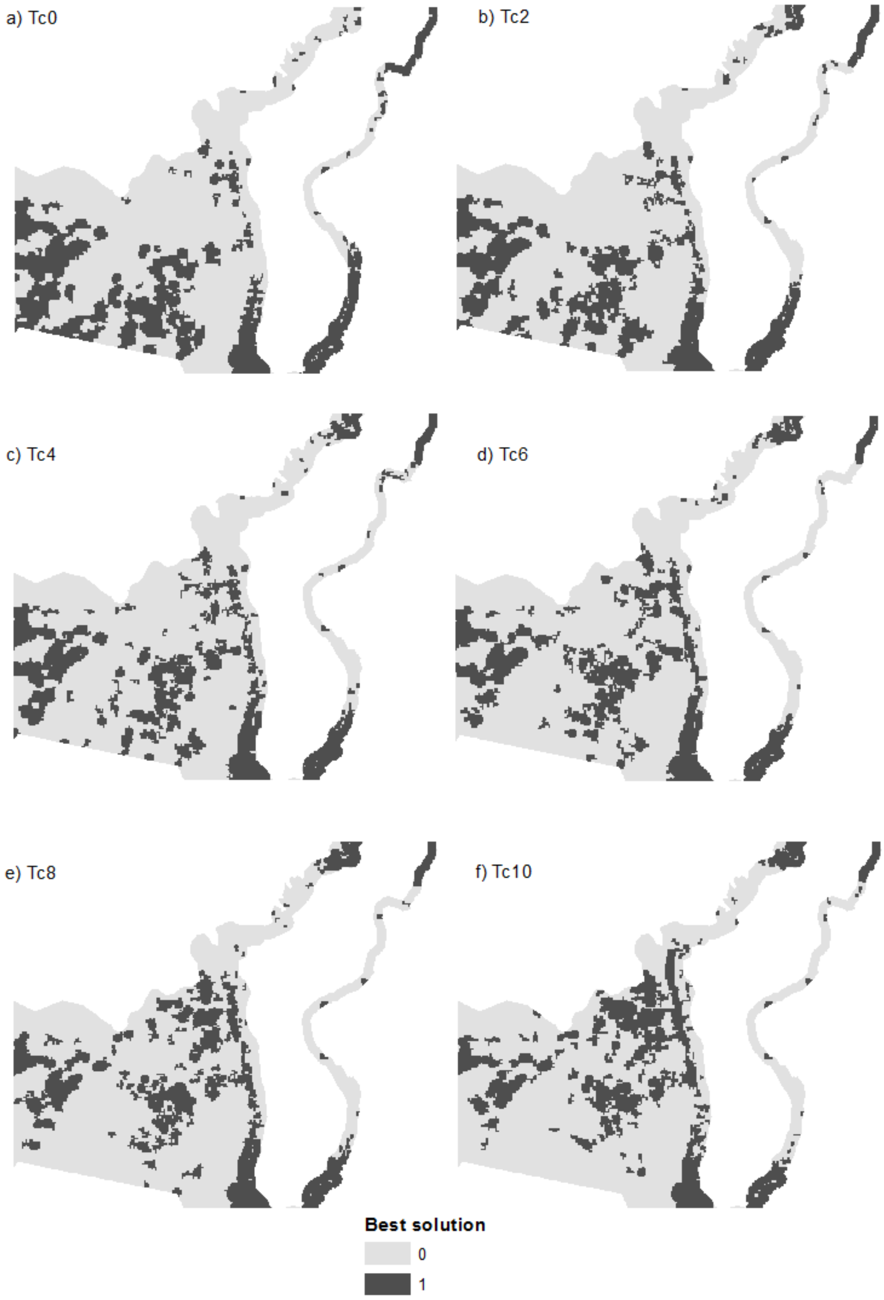


Figure 8.5. Best solutions produced by MinPatch using high biodiversity targets for each of six cost layers. Areas not selected in the reserve design are shown in light grey (0); areas selected for reserves are shown in dark grey (1). a) Tc0 (alpha=0), b) Tc2 (alpha=0.2), c) Tc4 (alpha=0.4), d) Tc6 (alpha=0.6), e) Tc8 (alpha=0.8), and f) Tc10 (alpha=1).

Table 8.5. The extent of overlap of planning units selected within the best reserve network solutions using different total cost layers. The Kappa statistic indicates a fair (0.21 to 0.4) to substantial (0.61 to 0.8) level of agreement between cost scenarios. Bold values highlight the total cost layer where alpha=0.6, which shows moderate to substantial levels of agreement across all scenarios.

Cost scenarios	**Tc0**	**Tc2**	**Tc4**	**Tc6**	**Tc8**
High targets					
Tc2	0.73				
Tc4	0.64	0.71			
Tc6	**0.55**	**0.65**	**0.69**		
Tc8	0.45	0.55	0.63	**0.70**	
Tc10	0.36	0.44	0.53	**0.61**	0.69
Low targets					
Tc2	0.73				
Tc4	0.63	0.74			
Tc6	**0.56**	**0.67**	**0.74**		
Tc8	0.44	0.55	0.61	**0.68**	
Tc10	0.33	0.42	0.50	**0.58**	0.69

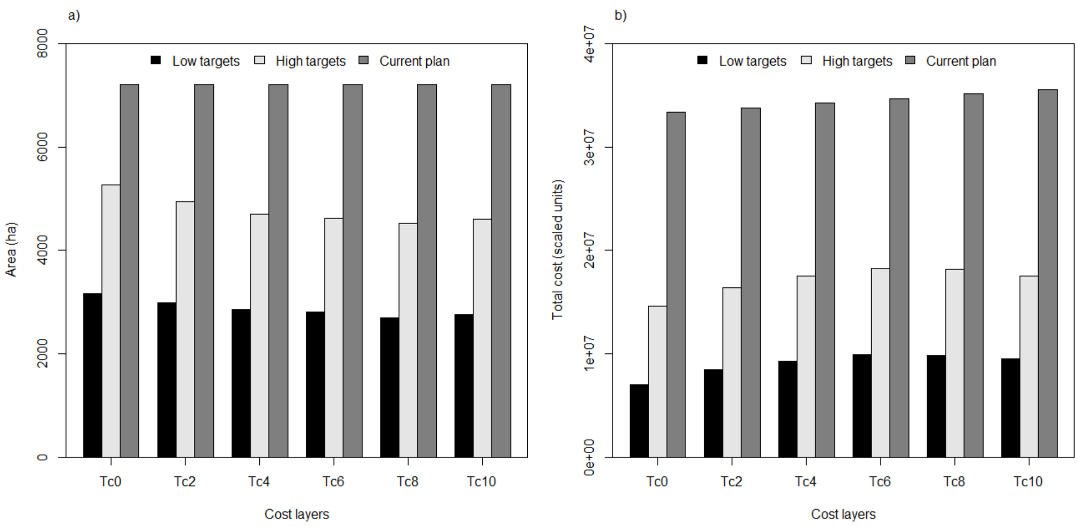


Figure 8.6. Total area (a) and cost (b) for the best solutions for low (black) and high (light grey) target scenarios for each of the six cost layers, and for the current management plan (dark grey). In all 12 scenarios, reserve area and cost for the current management plan greatly exceed those for the best reserve solutions found by Marxan and MinPatch.

8.3.3 Identifying priority areas

Selection frequency maps, also known as irreplaceability maps, can be used to identify priority areas for conservation (Stewart and Possingham, 2005). Marxan and MinPatch selection frequency maps represent the number of times a planning unit was selected out of the 500 runs for a given scenario (Figure 8.7). The spatial correlation of selection frequencies between high and low biodiversity targets was significant (Spearman's $r > 0.95$, $p < 0.0001$), meaning that scenarios with low targets generally selected the same planning units as the scenarios with high targets, and signifies the very clear prioritisation provided by the different biodiversity layers. As noted above, there is little variation across cost scenarios and strong positive correlations existed between selection frequency maps of different cost layers (Table 8.6). Planning units that were essential or unimportant for meeting targets maintained high or low priorities, respectively, regardless of the cost layer used. However, planning units of intermediate importance provided some flexibility for achieving targets, which could be exploited when reaching compromises during a participatory planning process. Importantly, the similarity of selection frequencies and variation across the planning area (i.e., some planning units were never selected while others were frequently selected) indicates that selection is not driven by chance but by biodiversity features and targets. As shown by Figures 8.1 and 8.2, the planning units most frequently selected align closely with highly suitable lemur habitat as well as higher quality vegetation. Furthermore, sites where the edge zone and good quality marsh overlap to encompass all three biodiversity layers were selected with especially high frequency.

Table 8.6. MinPatch selection frequencies between cost scenarios for high and low biodiversity targets were compared using Spearman's rank correlation coefficient. Correlations weakened with greater difference in total cost layers (i.e., from Tc0 to Tc10). All correlations had a significance level of $p < 0.0001$.

Cost scenarios	**Tc0**	**Tc2**	**Tc4**	**Tc6**	**Tc8**
High targets					
Tc2	0.95				
Tc4	0.88	0.96			
Tc6	0.78	0.89	0.96		
Tc8	0.67	0.80	0.90	0.97	
Tc10	0.55	0.69	0.81	0.91	0.96
Low targets					
Tc2	0.92				
Tc4	0.82	0.92			
Tc6	0.69	0.82	0.92		
Tc8	0.54	0.68	0.82	0.93	
Tc10	0.39	0.53	0.68	0.83	0.93

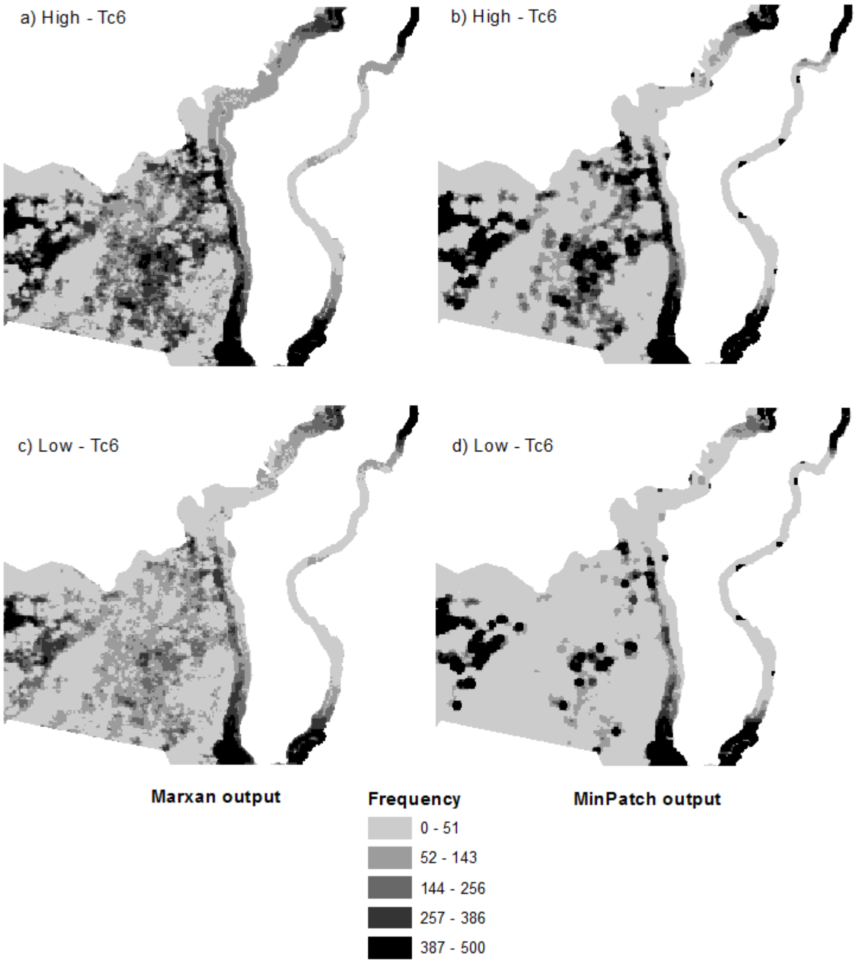


Figure 8.7. Marxan and MinPatch selection frequency maps using total cost layer Tc6 where alpha=0.6. Selection frequency is the number of times a planning unit was selected out of the 500 runs for each scenario; from low (light grey) to high (black) frequency. a) Marxan high biodiversity targets, b) MinPatch high biodiversity targets, c) Marxan low biodiversity targets, and d) MinPatch low biodiversity targets.

8.3.4 Comparisons with the currently proposed reserve design

Analysis of the proposed reserve design in the current management plan in terms of target achievement and total cost confirmed two key results. First, while proposed spatial restrictions in the current plan are close to several of the post hoc biodiversity targets that I set, such as for lemur habitat and higher quality reeds, the plan has a disproportionately high representation of low quality reeds (Table 8.7). The plan also conserves large areas of undesirable or untargeted features, such as invasive aquatic plants, savanna habitat, and open water, which are not reserved in any of the Marxan-MinPatch scenarios. Furthermore, although the current management plan reserves less edge area, this is clumped in very large and unevenly distributed blocks; only 68% (23 of 34) of edge zone blocks contained reserves.

Table 8.7. Extent of target achievement and area of biodiversity features conserved with the current management plan. High and low targets are the post hoc percentage targets I set (see Table 8.4) represented in terms of total area. Target achievement indicates under- (<100%) or over-achievement (>100%) of the high and low targets by the current management plan. Only 23 of the 34 (68%) edge zone blocks contained a minimum of 4ha in each.

				Current management plan		
Biodiversity feature	**Total area (ha)**	**High target (ha)**	**Low target (ha)**	**Area conserved (ha)**	**% High target achieved**	**% Low target achieved**
High quality reeds	627.7	439.4	313.9	310.7	71	99
Low quality reeds	6,037.3	603.7	301.9	2,437.8	404	808
Medium quality reeds	2,333.3	1,166.6	700.0	1,393.8	119	199
Phragmites	41.7	12.5	8.3	12.3	98	147
Lemur habitat	2,861.7	1,430.9	858.5	1,579.8	110	184
Nurseries / edge	4,345.4	1,738.2	1,303.6	1,245.9	72	96
Edge zone blocks (34)	-	Min 4 each	Min 4 each	-	68	68

Second, the total reserve costs and areas for the current management plan greatly exceeded those for the best reserve solutions found by Marxan and MinPatch (see Figure 8.6 above). Across the range of cost layers, costs for the current plan were from 90% to 128% greater than the best solutions for scenarios with high biodiversity targets, and from 249% to 375% greater than the best solutions for scenarios with low biodiversity targets. In addition, the current plan encompasses 7,213ha of conserved area, which represents 41% of the total planning area. The results in Table 8.7 and

Figure 8.6b show that the current plan does not meet all of the high biodiversity targets and costs considerably more than the best solutions found by Marxan and MinPatch. Furthermore, planning units reserved under the current management plan were significantly more costly, on average, than planning units selected by Marxan and MinPatch (Wilcoxon signed rank test, V = 81990415, $p < 0.0001$); also planning units not reserved under the current management plan were significantly less costly, on average, than planning units not selected by Marxan and MinPatch (Wilcoxon signed rank test, V = 346094895, $p < 0.0001$) (Figure 8.8).

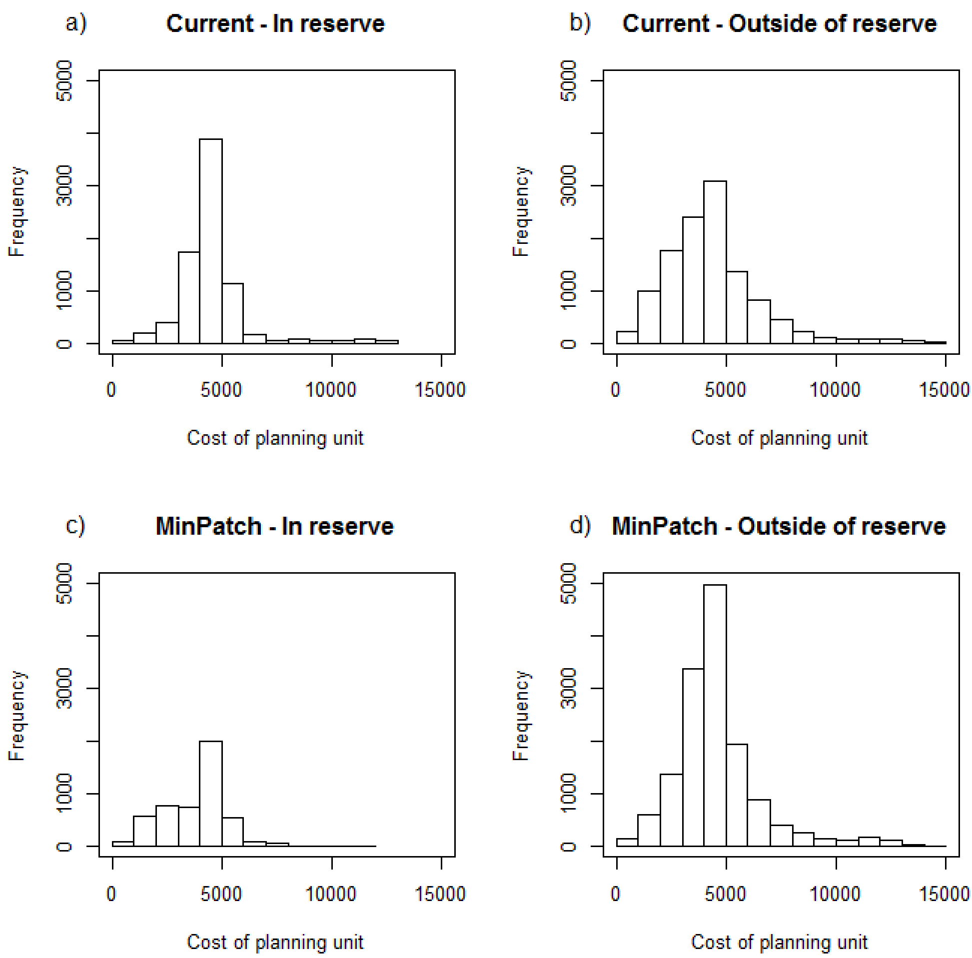


Figure 8.8. Frequency distribution of the costs of planning units inside and outside the current management plan (a and b, respectively) compared to planning units inside and outside the reserve for the best MinPatch solution using high biodiversity targets and total cost layer Tc6 (c and d, respectively).

In contrast to the reserve design under the current management plan, biodiversity targets were achieved for all scenarios run through Marxan and MinPatch. Although targets were consistently over-achieved for low quality reeds (range 161% to 245% across scenarios) and phragmites (range 187% to 269% across scenarios), this was probably because of a) the large proportion of low quality reeds in the planning area and b) the scattered distribution of phragmites amongst high quality reeds (also high quality lemur habitat) and medium quality reeds, both coupled with c) the minimum reserve size constraint set for the marsh zone. This additional constraint probably caused selection of planning units with these types of vegetation to ensure that reserve patches in the marsh met the minimum size requirement. Nevertheless, over-achievement of targets for low quality reeds in Marxan-MinPatch solutions was considerably less than for the current plan. Despite the over-achievements, the more efficient Marxan-MinPatch solution with alpha=0.6, where high targets are met and costs are minimised, encompasses 4,623ha (36% less than the current plan) or only 26% of the total planning area (Figure 8.9).

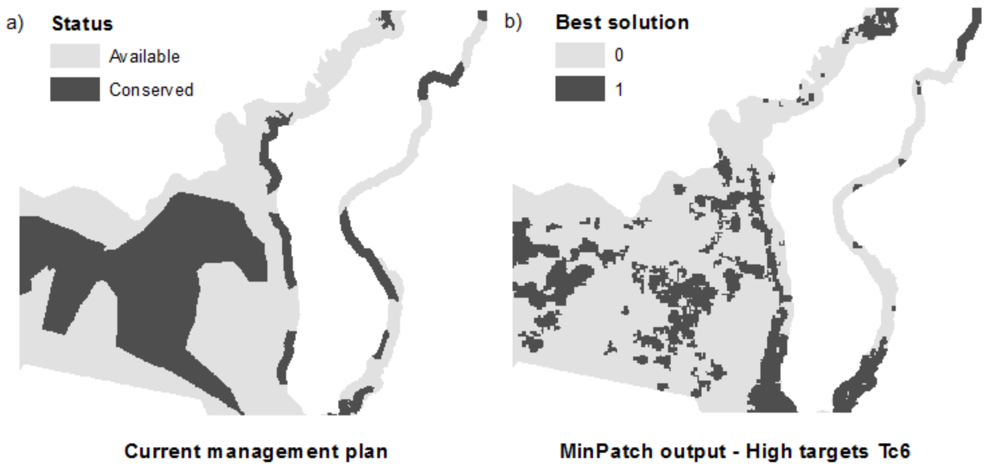


Figure 8.9. Spatial configuration of a) the current management plan compared to b) the best MinPatch solution using high biodiversity targets and total cost layer Tc6. Areas not selected in the reserve design are shown in light grey (Available or 0); areas selected for reserves are shown in dark grey (Conserved or 1).

8.4 DISCUSSION

Reserve systems worldwide are commonly acknowledged to be inefficient (Pressey and Tully, 1994; Fuller et al., 2010). In response, systematic conservation planning has developed rapidly over the past decade as a framework for identifying priority areas for conservation action (Margules and Pressey, 2000; Margules and Sarkar, 2007). However, the costs of conservation for resource users have rarely been quantified in a meaningful way during the planning process (see Naidoo et al., 2006; Carwardine et al., 2010). Costs for resource users are inherently socioeconomic; however, socioeconomic (as well as ecological) information is typically lacking in developing countries, particularly at spatial scales relevant to conservation planning (Ban et al., 2009; Weeks et al., 2010c). This case study shows that fine-scale catch and socioeconomic data collected from fishers can be used to represent spatially-explicit costs for fishers. In contrast with previous studies concluding that cost drives reserve design (Bode et al., 2008a; Adams et al., 2010), the planning units most frequently selected for reserves in the Lake Alaotra wetland are closely linked with biodiversity layers (lemur habitat suitability and vegetation quality). Lemur habitat and medium to high quality reeds were more likely to be selected than the remainder of the planning area. Planning units that were essential or unimportant for meeting targets maintained high or low priorities, respectively, in all scenarios regardless of cost layer used. However, reserve design was sensitive to how costs were weighted, with alternative cost layers resulting in key differences in reserve placement, particularly near the focal village.

The two forms of cost used in this study represented the relative economic values of planning units based on catch weight (catch cost) and the relative costs of accessing planning units in terms of distance from the village (proximity cost) where the cost of placing reserves close to the village is high. Accounting for spatial variation in costs minimises conflict that arises when protected areas restrict resource users excessively (Ban and Klein, 2009; Weeks et al., 2010c). While interviews suggested that catch is likely to be more important than travel in influencing fisher behaviour (see Chapter 6), the relative importance of the two forms of cost should be elicited explicitly from fishers and other stakeholders to determine which costs take precedence, and how they should be represented and weighted to find the best solution.

Although the current Lake Alaotra management plan generally meets the high biodiversity targets specified, it also covers large areas of untargeted habitat; this potentially makes compliance costly and enforcement challenging, which could result in a 'paper park'. While the implicit goal of protecting as much continuous marsh habitat as possible for the Alaotran gentle lemur is admirable, this might increase adverse impacts on fishers and other stakeholders. This outcome is inevitable when conservation and social goals are ill-defined (Adams et al., 2011). Furthermore, *ad hoc* reservation is spatially inefficient, fails to provide adequate protection where it is needed most, and can lead to high opportunity costs that compromise effective biodiversity conservation (Pressey and Tully, 1994; Stewart et al., 2003).

Despite the benefits of transparency and repeatability, Marxan (like most other planning tools) relies on static information, particularly in relation to distribution of biodiversity (Ardron et al., 2010). Wetlands are highly dynamic systems, and Lake Alaotra has high inter-annual as well as intra-annual variability in water level and floodplain extent (Ferry et al., 2009). Extent of wetland and the distribution of its biodiversity are also variable. However, two of the three biodiversity layers used in this study incorporated the same 2007 Landsat imagery, which does not capture the dynamic nature of the system. Similarly, the cost layers developed and used in these analyses are static. I also did not incorporate any measure of potential benefits from establishing no-take zones that could enhance fish abundance and catch from spill-over or indirect effects. These benefits are, however, unlikely to materialise in the short-term (Russ et al., 2005), and immediate costs to fishers will have the greatest impact on the implementation and ongoing success of a conservation plan that restricts fisher behaviour.

My study does not account for human behaviour and fisher responses to changing conditions or reserve implementation. Fisher responses could lead to either more or less severe impacts on fishers, depending on adaptive changes in fisher behaviour and fisher distribution following reserve implementation (Salas and Gaertner, 2004; Fulton et al., 2011). Fisher responses could be incorporated into this planning framework by modelling human behaviour and exploring subsequent changes in fisher density and effort as well as catch size (Smith and Wilen, 2003; Moustakas et al., 2006; Milner-Gulland, 2012). Accordingly, a multi-year dynamic approach to designing a reserve network for the Lake Alaotra wetland may be more realistic. Temporal dynamics can be included in Marxan analyses by making a planning unit a place in time and space rather than only a place in space (Grantham et al., 2008). By collecting detailed catch data from key fishing villages around Lake Alaotra across seasons and over multiple years, similar reserve designs could be developed on a seasonal basis and may suggest alternative plans for different times of year allowing for seasonal or temporary reserves. However, the benefits of these scenarios would need to be considered in terms of the costs and difficulties to implement such designs (Lee, 1999; Grantham et al., 2008).

The analyses performed in this study were relatively simple, with few biodiversity layers and two forms of cost. Although the simplicity matched the management objectives and made the results easier to interpret, greater insight into the system would be obtained from more data regarding the fine-scale distribution of fishers from other areas, other biodiversity layers such as for amphibians or birds (currently only available at coarser scales, e.g., Kremen et al. (2008) and Milligan (2009)), as well as threats including agricultural run-off, habitat destruction (marsh cutting and/or burning), and sedimentation from deforestation (Pidgeon, 1996). While this study aimed to design a network of reserves to address habitat destruction and overfishing, conservation interventions should ideally consider other land- and water-based threats where they exist (Klein et al., 2010). Upstream land uses and catchment disturbances also impact freshwater ecosystems and are increasingly incorporated into freshwater conservation planning (Linke et al., 2011; Linke et al., 2012).

The main data restriction for my cost model is the lack of detailed catch and socioeconomic information for fishers from other villages. To reduce any mismatch between planning and implementation, fine-scale catch data from each of the 25 main fishing villages around the lake and marsh are required in order to develop a meaningful cost layer for the entire Lake Alaotra wetland (see Knight et al., 2008; Weeks et al., 2010b); however, the challenge is how to obtain those data reliably and cost-effectively on an ongoing basis. Local involvement in the planning process and ongoing monitoring is increasingly recognised as critical for the long-term success of any resource management plan (Danielsen et al., 2005; Ostrom, 2009; Pressey and Bottrill, 2009). Unlike in commercial fisheries where confidentiality issues often limit access to data (Hinz et al., 2012), small-scale artisanal fishers at Lake Alaotra are well-placed and willing to collect spatially-explicit catch data to improve fishery management. Using the uncomplicated datasheet designed for this study (see Appendix S3.1), fishers from each village could record their fishing activity on one specific day each month. This participatory approach to collecting data for spatial planning is not only reliable and cost-effective, but could also be expected to foster greater stakeholder involvement and collaboration, planning decisions that are more comprehensive, and higher levels of compliance; it could also lead to suggestions for developing an implementation strategy (see Knight et al., 2006b). Participatory monitoring is one way of promoting crucial local buy-in to the planning process over the long term. Monitoring programs to test the effectiveness of interventions should be an integral part of planning (Powers and Abeare, 2009).

Using Marxan and MinPatch to generate a new reserve plan is only one step in the structured process of iterative, stakeholder-driven workshops and planning necessary to design an efficient, transparent, and participatory management plan that will be implementable and effective for biodiversity conservation (Knight et al., 2006b). It should be noted that a fragmented distribution of restricted areas, as developed in this study within marsh habitat, might not be feasible in practice due to undesirable edge effects and associated difficulties of demarcation and enforcement. Once a reserve plan has been drafted, a consultation process should follow to give local resource users and other stakeholders an opportunity to provide input (see Adams et al., 2011; Jupiter and Egli, 2011). Setting aside time for communities to respond to a proposed plan would help cultivate institutional relationships while fostering community engagement in the planning process and negotiation of the best solution (Game et al., 2011; Grantham et al., 2012). The consultation process in this case could also be used to elicit information regarding the relative importance of the two forms of cost, identifying further forms of cost, and as a type of social marketing to shift resource-user behaviour in favour of biodiversity conservation (Kotler and Zaltman, 1971; McKenzie-Mohr and Smith, 1999).

Qualitative goals, expert opinion, and information from local resource users are important for shaping more-specific, quantitative objectives (such as targets) that are essential to make conservation management effective, repeatable, and transparent (Knight et al., 2006a). Anororo-based fishers generally perceived inequities in the distribution of costs and benefits under the current management plan, which may lead to social or political conflict, failure during implementation, or poor compliance (see Cinner, 2007; Klein et al., 2009). The Marxan-MinPatch solutions developed in this study could reduce conflict by producing socially acceptable configurations based on well-defined targets and constraints that consider the needs of resource users.

Consideration of costs for resource users is critical for sustainable fisheries management and effective long-term conservation of freshwater ecosystems. The challenge is to develop a management plan for protected areas and no-take zones that balances biodiversity conservation with the needs of local people. The process of incorporating socioeconomic data into conservation planning is integral to ensuring management plans do not have adverse or unexpected outcomes (Muller and Albers, 2004) and will maximise the probability of success. My research demonstrates an effective approach for incorporating fine-scale socioeconomic data into planning for biodiversity conservation and fisheries management at local scale, and shows that this integration can indeed lead to spatially-efficient reserve designs that are substantially less costly for resource users.

CHAPTER 9 DISCUSSION

9.1 SYNTHESIS AND CONTRIBUTIONS OF RESEARCH FINDINGS

The research reported in this thesis is embedded within the contextual frameworks of systematic conservation planning and fisheries management. Following recent trends for research within these fields to afford greater attention to socioeconomic factors (for example, Abernethy et al., 2007; Ban and Klein, 2009; Weeks et al., 2010a; Linke et al., 2011), the approach and focus of the study is from the perspective of fishers as vested resource users. This builds upon relatively recent shifts in research priorities from understanding fisheries' resources to understanding the users of those resources (Abernethy, 2010).

Research addressing fishery management and conservation objectives has historically been most concerned with fish ecology and biological processes within fisheries, and then proscribing fishers from making levels of catches calculated to be unsustainable on the basis of stock assessments (see Munro, 1983; Hilborn and Walters, 1992). This primarily top-down biological, quantitative approach to understanding and managing fisheries has delivered few successes but many disappointments or failures, especially in small-scale fisheries (Pitcher et al., 1998; Berkes, 2003; McClanahan and Castilla, 2007). The top-down approach has also failed to account for variation in livelihood strategies used by fishers (Allison and Ellis, 2001). To improve outcomes, my study continues the growth of crucial research incorporating social dimensions into fisheries management and acknowledging the role of fishers as influential active constituents within social-ecological systems (see Berkes, 2004; Charles et al., 2009; Coulthard et al., 2011).

To address fishery management and associated wetland conservation issues it is first essential to have a detailed understanding of the system and context within which the issues occur. The characterisation of Lake Alaotra's fishery provided in my study is the first in more than three decades and the most comprehensive (Chapter 4). Because the Alaotra wetland is a dynamic system that has varied considerably over time and requires an adaptive approach to management, my characterisation is not only timely but directly of value to organisations such as Durrell Wildlife Conservation Trust, the

Department of Water and Forests, and the regional Fisheries Service charged with managing and conserving the system. A major part of the challenge of managing the fishery is the long history of mismanagement through introductions of exotic fish species as well as poor governance; systematic efforts to manage the fishery for livelihood sustainability in conjunction with biodiversity and wetland conservation are recent (Durbin et al., 2003; Ranaivonasy et al., 2005).

Key findings from characterising the fishery are that:

- many people depend highly on the fishery for livelihood throughout the year;
- there is considerable unpredictability and variability in catches;
- Anororo-based fishers use a broad range of fishing locations and methods;
- fisher behaviour varies over the year in an effort to maintain catch;
- catch composition varies across habitats and between seasons;
- daily and seasonal income from fishing is highly variable between fishers;
- a high proportion of fish are caught before reaching maturity; and
- fishers perceive the fishery to be in a state of decline.

Water level was identified as a key determinant of choice of fishing location and/or type of gear used, as well as the spatial distribution of fishing activity over the year. Anororo's subsistence and cash economy depends on fishing and rice cultivation, and there are few alternative livelihoods. With a currently increasing human population, limited land for rice fields, growing fisher numbers and fishing effort, and declining catches and fish sizes, the fishery and adjacent wetland habitat is under increasing pressure. This is typical of many freshwater artisanal fisheries in developing countries (see Durand, 1979; Amarasinghe and de Silva, 1999; Balirwa et al., 2003; Béné et al., 2003; Matsuishi et al., 2006), highlighting the need for insights to better understand such fisheries and improve their management and conservation. My characterisation of Lake Alaotra's fishery contributes to this by providing solid foundations for more complex analyses to assess the drivers of catch size and fisher spatial behaviour, the costs to fishers of alternative management interventions, and tradeoffs and synergies between fishers' interests and conservation goals.

The short-term costs of management interventions for resource users (Naidoo et al., 2006; Daw et al., 2011a) and their impacts on user behaviour and livelihood are often unclear or rarely specified in fisheries and conservation research (Cinner et al., 2008; Holland and Herrera, 2009). My study addresses this shortfall by explicitly exploring the potential impacts of interventions on fishers, identifying direct short-term costs to fishers through increased effort and reduced catch sizes, and demonstrating that the costs vary between groups of fishers according to the type of gear used (Chapter 5). The finding that interventions can impact resource users unevenly contributes to potential fine tuning of management and conservation strategies to account more-precisely for costs to key or specific resource users.

The research makes an important contribution to our understanding of the dynamics of artisanal (and other) fisheries by evaluating and explaining fisher spatial behaviour at a scale relevant to conservation planning (Chapter 6). Studies have historically focused on large-scale commercial fisheries, usually within developed countries, when conceptualising and characterising fishers' decision making and patterns of behaviour (see Abernethy, 2010; Tidd et al., 2011). Information from studies of large-scale commercial fisheries has typically been inadequate or inaccurate when applied to small-scale artisanal and subsistence fisheries within developing countries, leading frequently to misunderstandings about fishers' motivations, fishing objectives, and behaviour (Hilborn, 2007). These misunderstandings have in turn often led to inappropriate and ineffective management interventions, or compromised compliance with regulations by imposing greater costs on fishers than they are able to bear (Peterson and Stead, 2011). My findings support the view that the costs of management actions are likely to have substantial impact in subsistence, artisanal, and developing-country settings because fishers will typically be socioeconomically highly invested in and dependent on fishing. These fishers will also probably be relatively poor (and hence vulnerable and less resilient to shocks), lack buffers to offset seasonal variation in catches and income, and have limited livelihood options (Hill, 2011). Accordingly, future research should continue to investigate the nuances and contextual specificity of fishers' decision making and spatial behaviour in order to recommend management actions and conservation interventions with minimal negative impacts for fishers who depend on the fishery for livelihood. This could in turn increase fisher compliance and hence the effectiveness of such actions and interventions.

Because the resources of most freshwater artisanal fisheries are heavily exploited (FAO, 2010; Welcomme, 2011b) there is an increasing need for active fisheries management to secure sustainable resource extraction over the long term, as well as conservation benefits for adjacent wetlands and biodiversity. Spatial restrictions on fishing and/or limits on fishing effort are increasingly implemented to address these issues, given that aquatic protected areas can increase fish stocks, reduce environmental impacts from fishing, and increase catches in adjoining areas (Roberts et al., 2001; Gell and Roberts, 2003; Suski and Cooke, 2007). The efficacy of such interventions is likely to hinge on fishers' willingness and ability to comply, as well as how fishers redistribute their effort outside restricted areas. Benefits arising from spatial restrictions could be overridden by the adverse impacts of increased fishing intensity if greater numbers of fishers aggregate into smaller fishing areas, even over the short term. A primary finding of the study is that fishers' decisions regarding compliance and effort distribution, and therefore the efficacy of many management and conservation actions, ultimately depend on the actual and perceived costs of the actions for fishers (Chapters 6 and 7). These costs include tradeoffs with interests and objectives other than catch that can also drive fishers' behaviour. Greater research attention should be afforded to understanding the relationships between fishers' motivations, perceptions of the costs of management actions, and responses to those actions.

My research also draws attention to the often overlooked or understated tendency of fishers' spatial behaviour to be relatively site- or context-specific, and tied to factors such as diversity of habitat within the fishery and type of fishing gear used. At broad scale, drivers of the behaviour of Anororo-based fishers were consistent with drivers in other fisheries and mainly linked to individual motivations, livelihood objectives, and fishers' social interests (Chapter 6; Béné and Tewfik, 2001; Cabrera and Defeo, 2001; Salas and Gaertner, 2004; Daw, 2008). At finer scale, type of fishing gear used and habit were the key factors driving effort and choice of fishing location for Anororo-based fishers (Chapter 6). Gear use is interlinked with type of habitat at fishing locations, and fishers largely perceive their catches to be linked to expertise with gear and familiarity with location. Analysis of fisher behaviour should therefore occur at local level, especially because this is where fisheries management and conservation can be addressed most effectively in artisanal contexts.

Fishers and other resource users are frequently conceptualised as rational utility and profit maximisers, portrayed as *homo economicus* when making livelihood decisions. Traditional microeconomic theory conceives *homo economicus* as the operant agent in economic activities. Although primarily a modelling construct to aid analysis (Persky, 1995) he/she is the standard economic conceptualisation of human behaviour, which focuses on evaluating choices, making decisions, and resolving strategy efficiently in market exchanges (Morse, 1997; Güth, 2008). The allied theory of rational expectations (see Muth, 1961; Lucas Jr., 1972) is based upon the assumption that agents have full knowledge of markets and use this to form expectations about how their decisions will affect their wellbeing or utility (Balakrishnan et al., 2000; Edwards-Jones, 2006). *Homo economicus* always acts rationally, exhibits highly stable preferences which are similar across individuals, responds systematically to market constraints, and invariably makes optimal choices to maximise personal utility (Stigler and Becker, 1977; Thaler, 1988; Vriend, 1996). Although utility or profit maximisation (also preference maximisation: Gintis, 2007) is largely circular because it can only be inferred from behaviour and outcomes, it has historically been a fundamental concept within fisheries research (Daw et al., 2011b); fishers are deemed to behave as *homo economicus* when making fishing decisions and management has been designed on this basis, often with ineffective results (Hilborn, 1985; Salas and Gaertner, 2004; Holland, 2008).

My research indicates that it is simplistic and inaccurate to characterise all fishers as utility and profit maximisers. Anororo-based fishers valued routine and convenience, and distributed their effort to where they perceived they would have consistent and sufficient (but not necessarily maximal) catches after accounting for their broader personal and socioeconomic interests (Chapter 6). Fishers' decisions about effort allocation were often tied to unique environmental or anthropogenic features of fishing sites, which were important for gear-specific reasons. By demonstrating that fishers' behaviour is typically mediated by tradeoffs and fishers do not always fish in ways to maximise returns, my study contributes to a richer understanding of the complexities of fishers' decision making. This understanding could be used to advantage in management and conservation to ensure plans and actions accommodate resource-users' behaviour; it could also inform the development of meaningful and realistic incentives to support interventions and comply with regulations.

My findings regarding fishers' behaviour and decision making suggest that artisanal fishers are not only boundedly rational (Jones, 1999; Gintis, 2007) but also typically risk averse (see Kahneman and Tversky, 1979; Kahneman, 2011); under conditions of uncertainty they satisfice with sub-maximal catch instead of optimising their returns (Simon, 1955; Foxon, 2006). Absence of advanced technology heightens uncertainty in artisanal fisheries, and despite drawing upon traditional knowledge and heuristics many fishers opt to maintain the *status quo* to access a hidden resource. My research shows that prospects of better catches and/or lower costs might have relatively little effect on changing an individual fisher's behaviour due to the constraining influence of tradeoffs important to them (Chapter 6). A key implication of this finding for applied research and management is that fishers may be unresponsive to purely-economic incentives to modify their behaviour, or less responsive than commonly expected.

Fishers' perceptions and anticipated responses to change provided insights about the probable impacts of management interventions for fishers, which suggests that these types of data should enhance managers' ability to design interventions for high levels of compliance. Because Anororo-based fishers share similar levels of awareness and perceptions about the current fishing regulations, it is likely that infringements arise predominantly as a function of compliance costs (Chapter 7). For example, most fishers continue to sell fish during the closed period due to the economic hardship it causes. Variation in fishers' responses to scenarios highlighted individual differences in objectives, social priorities, and livelihood options, which affirms the value of employing social science methods in fisheries research. My findings for fishers' adaptive responses show that context is fundamental to assessing the effects of amplifying or dampening fishing effort. Also, displaced effort due to spatial management interventions is unlikely to simply dissipate into alternative economic activities (where available); rather, it may intensify problems and engender conflict in other locations. The implication is that management must address these issues directly instead of avoiding them, and also be adaptable in approach because each intervention will typically have knock-on effects.

A highly important finding of the research for fisheries and conservation governance is that fishers were generally in favour of regulations and management interventions, and perceived that an absence of regulations could lead rapidly to fishery depletion or collapse (Chapter 7). Fishers not only understand clearly the direct connection between condition of the fishery and livelihood security and returns, but also want to have regulations enforced to ensure fishery sustainability and their livelihood over the long term. It is often assumed that resource users such as fishers inherently overexploit open-access resources in the manner of the ‘tragedy of the commons’ (Hardin, 1968; Berkes, 2003); however, my study confirms that this is not necessarily the case. Many fishers felt disappointed and let down by governance institutions, and would like authorities to participate in the fishery according to their mandates.

An additional key finding was that Anororo-based fishers are willing to engage with governance institutions and positive about the prospects for comanaging the fishery (Chapter 7). This is encouraging because management interventions are more likely to be viewed favourably by fishers when designed and introduced in collaboration with them (Hart and Pitcher, 1998; Salas and Gaertner, 2004). Despite potentially adverse short-term impacts from interventions, fishers are keen to develop sound strategies for fishery sustainability, based on robust science as well as consultation. Similarly, fishers are generally aware of the main problems in the fishery and would contribute to solutions if their interests and costs are taken into account. The analyses of fisher perceptions and behaviour undertaken in my research suggest ways for this to occur.

Margules and Pressey (2000) developed a framework for conservation planning that has been extended by Knight et al. (2006a) and Pressey and Bottrill (2009). Each representation of the framework has multiple stages to determine when, where, and how conservation actions can be implemented and monitored most effectively. Other representations of the planning framework vary only in terms of the components and interrelationships highlighted in a particular case (Pressey and Bottrill, 2009). Figure 9.1 depicts the stages of the planning process, including where socioeconomic data and/or information from monitoring, as collected in my research, can be integrated into this process. Despite the unidirectional nature of the diagram, some stages may occur simultaneously; there will also be numerous feedback loops to adjust earlier decisions, highlighting that the process is inherently adaptive.

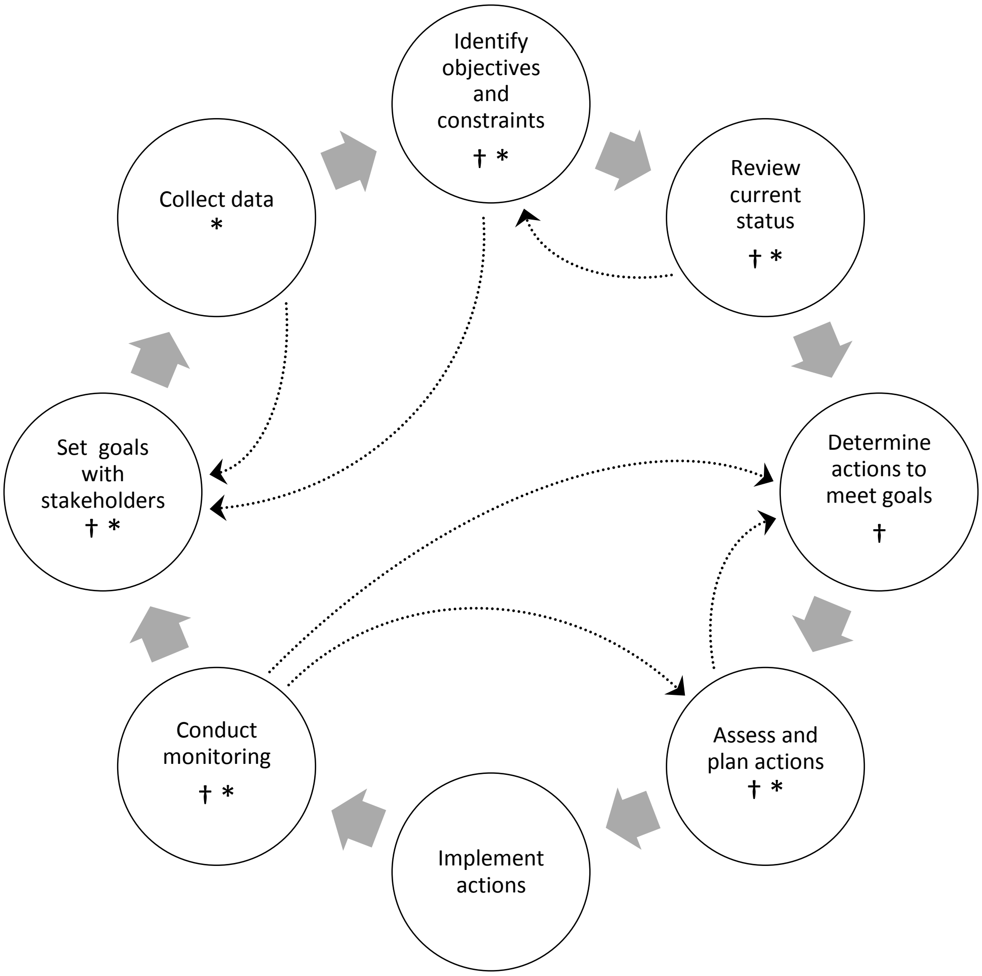


Figure 9.1. Stages in systematic conservation planning, adapted from Margules and Pressey (2000), Knight et al. (2006a), and Pressey and Bottrill (2009). Stages where socioeconomic data (†) and/or information from monitoring (*) collected in my study can be integrated into the planning process are shown.

My research increases our ability to design and configure effective reserve networks by demonstrating that fine-scale catch and socioeconomic data collected from fishers can be used to represent spatially-explicit costs for resource users within systematic conservation planning (Chapter 8). Because costs of management and conservation for resource users are often inaccurately or inadequately accounted for during the planning process (see Naidoo et al., 2006; Carwardine et al., 2010), this is a novel and important step forward. It not only allows planners to make reserve-design decisions that are locally appropriate but also draw upon data that can be readily collected using participatory approaches. My analysis using Marxan and MinPatch shows that it is feasible to achieve efficient reserve configurations for artisanal fisheries and associated wetlands that reduce costs to resource users, decrease potential conflict between resource users and governance institutions, and meet or exceed biodiversity targets. The analysis also highlights the utility of Marxan and MinPatch software as conservation planning tools. However, a highly fragmented distribution of restricted areas might generate undesirable edge effects that lead to difficulties in demarcation and enforcement. Further research is warranted to explore whether the reported gains in efficiency and reductions in costs to resource users are achievable in a network of more consolidated patches.

As noted by others (see Bernard, 2002; Randall and Koppenhaver, 2004; Redpath et al., 2004; Reed, 2008), gaining the trust and confidence of resource users is essential when seeking to understand their perceptions and behaviour. My research is further confirmation that engendering trust, confidence, and research credibility with resource users improves data quality and insights because they will be more open and honest when answering questions, and also more likely to allow a researcher to 'step inside' their daily way of life. This was achieved in numerous ways, including:

- living within the Anororo community while in Madagascar;
- taking part in events and activities that are part of village life;
- recruiting respected local assistants and key informants;
- being explicitly interested in understanding the fishery and fishers' perspectives in order to improve returns for fishers and facilitate more-equitable management;
- following fishers to observe and experience their fishing activities;
- being honest and consistent, and maintaining confidentiality and anonymity;

- not monitoring infringements of regulations, except for data collection;
- not policing fishers or reporting infringements to authorities;
- being reliable, doing things promised, and not building unrealistic expectations;
- spending time with fishers without collecting systematic data.

Although many of the types of behaviours listed above might appear to be common sense or taken as givens in research settings, in my experience they are infrequently practiced and rarely considered a core component of project methods; this can build walls to data and understanding, such that researchers fail to acquire the full picture. Many of the key insights in my study, especially in relation to scenarios, compliance or non-compliance with regulations, and costs to fishers, were underpinned by trust and confidence from fishers as well as community support. In this regard the research makes an invaluable applied contribution to project approach and design, particularly when personal, behavioural, and contentious or sensitive information is sought from resource users.

The research also confirms that integrated interdisciplinary approaches are not only highly applicable but essential when investigating fisheries management and wetland conservation issues. My analyses draw upon findings in ecology, biology, economics, behavioural economics, management, psychology, fisheries science, and conservation science. This breadth of perspective gives "a multidimensional view that incorporates ecological, socioeconomic, community and institutional arrangements in the ... system evaluation (and generates) ... a 'portfolio' of approaches to provide multidimensional solutions" to complex problems (Salas and Gaertner, 2004, p.163). Because fisheries are dynamic social-ecological systems (Berkes, 2004; Ostrom, 2009; van Poorten et al., 2011; Cinner et al., 2012), they can only be managed and conserved effectively when interactions between social and ecological components are understood; this is evident in results throughout this thesis.

Similarly, resource-management researchers increasingly recognise that multifaceted issues with interacting social, cultural, economic, and ecological dimensions (such as fisheries and wetland conservation) cannot be validly examined or addressed using a single method (see Grimble et al., 1995; Nyhus et al., 2002; Dudgeon et al., 2006).

My results are derived from analyses of data collected through a range of techniques, including structured interviews, semi-structured interviews, scenarios, focus groups, participant and direct observations, informal discussions, and existing records. This breadth of sources of information facilitated triangulation of data while contributing to a more-complete contextual understanding of fishing behaviour and socioeconomic factors, confirming the value of multi-method research in fisheries management and conservation (see Pound et al., 2003; Ormsby and Kaplin, 2005; Daw et al., 2011b).

9.2 FUTURE RESEARCH DIRECTIONS

Feasible research within the timeframe of a PhD program is necessarily constrained both spatially and temporally; it is limited to collecting data over a relatively short period of time and in one or a few locations. In this case I focused my research on the village of Anororo because it is adjacent to marsh and lake habitat, close to planned conservation interventions, has a large population of fishers using a variety of habitats and fishing methods, and there is a high local dependence on fishery resources for subsistence and commercial activity. This raises the question of the extent to which the research is representative of the lake as a whole and over time. *Ad hoc* meetings with fishers from other villages around Lake Alaotra confirmed that their views and concerns were equivalent to those of Anororo-based fishers. Further insight into the dynamics and functioning of the system would be gained by longer-term monitoring of the fine-scale distribution and perceptions of fishers from all areas of the lake. However, a framework by which this data contributes to understanding the system and informs management is required.

While this thesis filled essential data gaps regarding the status and functioning of a freshwater artisanal fishery in a developing country, and examined critical issues related to the human side of fisheries management, questions remain concerning the interactions and dynamics between management interventions and the adaptive responses of fishers. In particular, further research is needed to link the in-depth knowledge about fishers' adaptive responses collected during scenario interviews in this study with a new management plan in an explicit way. My research relies on interview responses to hypothetical scenarios to assess the impacts of different management interventions. However, these impacts could be more or less severe depending on actual adaptive changes in fisher behaviour and fisher distribution following implementation of interventions.

Management strategy evaluation (MSE) is a tool used to inform resource management and conservation that involves using simulation to evaluate alternate management strategies based on identified management objectives and measures of performance (Sainsbury et al., 2000). The method allows quantitative analysis of options for meeting the objectives of multiple stakeholders within a common, participatory framework (Mapstone et al., 2008; Holland and Herrera, 2009). MSE is explicitly geared to formulating decision rules and management strategies that account for uncertainty, and are robust to natural variation within social-ecological systems (Kell et al., 2005; Holland and Herrera, 2009; Milner-Gulland, 2012).

MSE incorporates a series of models to simulate management dynamics within a social-ecological system (SES): (i) an operating model that reflects the dynamics of the population of management concern, (ii) a sampling or observation model simulating data collection within the system, (iii) an assessment model comprised of performance indicators that will be used to form decision rules, and (iv) a management model that simulates outcomes for a range of management actions (Butterworth and Punt, 1999; Sainsbury et al., 2000; Holland and Herrera, 2009).

MSE is flexible, well-suited to adaptive management practices, clarifies the trade-offs involved in management actions, and allows all stakeholders to have input into strategy evaluation (Smith et al., 1999; Sainsbury et al., 2000; Mapstone et al., 2008; Needle, 2008). Effective use of MSE relies on accurate parameterisation of the operating model (Sainsbury et al., 2000). Although MSE can model resource-user responses to incentives and regulations, this has not often been done and there is considerable scope to include socioeconomic data more explicitly to explore the impact of cost-benefit decisions by resource users (Mapstone et al., 2008; Holland and Herrera, 2009; Milner-Gulland, 2012). Using MSE for Lake Alaotra would allow exploration of the impacts of fisher adaptive responses on the dynamics of the fishery as well as how that information will feed back into the governance system. Finalising and implementing a new management plan for Lake Alaotra could take this MSE modelling one step further and allow assessment of actual versus simulated impacts.

In addition, the Marxan analysis conducted for Chapter 8 does not consider different management zones or uses of the marsh. This was reasonable given the fisher focus of my research; from the perspective of fishers the marsh is either open or closed for them to use for fishing. However, there are other resource users (such as rice farmers and those who make marsh products) who rely on the Lake Alaotra wetland for livelihood and could also be impacted by management interventions. This indicates that further analysis incorporating multiple cost layers for multiple stakeholders (not only fishers) would be worthwhile when planning over the entire Lake Alaotra wetland. Marxan with Zones is a recent extension of the Marxan software that permits zoning to take into account multiple uses and hence differences in costs between uses (Watts et al., 2009). However, in order to conduct this type of analysis and to reduce any mismatch between planning and implementation, cost and socioeconomic data from other stakeholder groups across all areas of Lake Alaotra would be required.

While Chapter 8 aimed to design a network of reserves to address overfishing, conservation interventions should ideally consider other land- and water-based threats where they exist (Klein et al., 2010). Other threats to the Lake Alaotra wetland include agricultural run-off, habitat destruction (marsh cutting and/or burning), and sedimentation from deforestation (Pidgeon, 1996). Upstream land uses and catchment disturbances also impact freshwater ecosystems and are increasingly incorporated into freshwater conservation planning (Linke et al., 2011; Linke et al., 2012). Hydrological conditions affect density-dependent population dynamics of freshwater fish species (Halls and Welcomme, 2004). Generally, exploitable biomass is increased during flood conditions and with increasing flood duration (Kolding and van Zwieten, 2012). The timing of dam opening and closing for irrigated rice fields adjacent to Lake Alaotra could have implications for the fishery in terms of fish spawning behaviour, primary production in the system, and availability of critical habitat and warrants investigation. Simulating the hydrological conditions of Lake Alaotra could provide guidelines for managing or manipulating water levels to mitigate potentially negative impacts on the fishery (see Moreau, 1979a; Halls and Welcomme, 2004).

The role of marsh habitat in supporting adjacent fisheries is widely recognised (Welcomme, 2001). Continued conversion of marsh habitat into rice fields is of concern for the Lake Alaotra wetland and may exacerbate current fishery declines (see Barbier, 2003). It would be informative to investigate the possible impact of irrigation for rice fields on marsh and fish habitat, and to estimate the potentially substantial livelihood and welfare impacts of continued marsh conversion for communities at Lake Alaotra.

9.3 RECOMMENDATIONS FOR MANAGEMENT

The series of management interventions trialled in the Lake Alaotra wetland have not been successful and the fishery remains in decline (Chapter 4). One of the main reasons for this is the lack of consideration for and consultation with fishers. Despite Durrell’s good intentions to include local fishers and stakeholders, the approach used to date has failed to properly engage them for management planning and implementation as well as ongoing monitoring. Although fishers view Durrell as an ally, they have no confidence in the Federation of Fishers or the Fisheries Service and believe corruption is a major problem (Chapter 7). The Federation of Fishers collects funds via license fees and a levy on fish sold to fish collectors, which is meant to assist fishers during the closed period. These funds have been collected since 2005 but so far have not been used to benefit fishers. My observations and discussions with fishers suggested there is little or no monitoring of the fishery, regulations are not enforced, and there is no work or help for fishers over the closed period (Chapter 7). This appears to be largely due to lack of will in addition to lack of capacity. Despite this, Anororo fishers are still positive, and want to be involved and consulted for advice as well as to see that regulations for fishing activities are enforced (Chapter 7). Although my results indicate that almost all fishers participating in the research agreed generally with current fishing and conservation regulations to protect the fishery and associated habitat, the direct and indirect costs of interventions to fishers (Chapters 5 and 8), as well as drivers of fishers’ behaviour and their perceptions (Chapters 6 and 7) have not previously been considered.

One of my key findings was the large proportion of juvenile fish caught (Chapter 4). Given that fishers construct their own fishing equipment and the lack of monitoring and enforcement, fishers have been free to use progressively smaller mesh sizes each year to catch smaller and smaller fish. Fishers noted two fishing methods in use that are particularly damaging for the fishery because they catch very small fish; the '*ramangaoka*' (large seine net with 1cm mesh size) and the '*sitra*' (dip net made of mosquito netting with 3mm mesh size) (Chapter 7). Anororo-based fishers nominated the illegal *ramangaoka* as the most destructive of these methods, in terms of both damage to habitat and damage to or complete loss of fishing gear belonging to other fishers. However, the *sitra* (which is a legal method for catching Madagascar rainbowfish and mosquitofish) was identified as the most destructive method in terms of the quantity of juvenile Nile tilapia caught. Although enforcement of gear-related restrictions is difficult, it is imperative for these two gear types to ensure survival of juvenile fish.

Another important finding was the mismatch between current timing of the closed period and fish biology (Chapter 4). Although adjusting the timing of the closed period to provide better protection for spawning fish is plausible, my research indicates that this will have unequal impacts on different groups of fishers (Chapter 5). Furthermore, fishers generally viewed the closed period as an intervention with more negative impacts than others (Chapter 7). The main problem associated with the closed period for fishers is that there is no other work or income available. Any change in the timing of the closed period should be linked with a local work program to reduce the negative impacts on fisher livelihood. Based on my observations and discussions with fishers, some suggestions for work could include hiring fishers to (i) remove invasive plants, (ii) demarcate mutually agreed no-take zones, or (iii) plant trees on hills to the east of the lake to reduce erosion. The levy collected by the Federation of Fishers could be used to fund such a program, which would help to ensure member fishers have an alternative source of income during the closed period (as intended when the levy was implemented) and increase the likelihood of compliance.

In light of recent research suggesting that spatial restrictions are more effective than temporal restrictions (Grantham et al., 2008; Game et al., 2009) and also cause less socioeconomic hardship for fishers, Durrell's intention is to phase out Lake Alaotra's closed period in favour of no-take zones. Fishers are also more in favour of spatial closures than temporal closures (Chapter 7). The 2006 Lake Alaotra management plan includes 12 no-take zones (NTZs) around the lake edge; however, Anororo fishers indicated that these zones were too large and very inconveniently located at primary access points to the lake from villages. Results from my study suggest that a network of smaller NTZs would minimise costs to fishers, distribute costs and benefits more equitably around the lake, and be better received by fishers (Chapter 7). When establishing no-take zones it is also important to understand the biology and ecology of dominant species; for example, tilapias rely on specific nesting sites and nursery grounds. Anororo fishers often referred to '*lozokas*' – natural bays or inlets around the edge of the lake – as ideal locations for no-take zones. It would be informative to investigate whether these sites are used by tilapia for spawning and as nursery areas with a view to following fisher recommendations if this is the case.

Accounting for the costs of conservation interventions to different fisher groups is a first step in improving upon and developing a new management plan for the Lake Alaotra wetland (Chapter 5 and 8). However, another key component in developing an effective management plan is understanding how resource users will respond to interventions. A disconcerting finding in my study was that fishers stated they would amplify their fishing activity under improving conditions (such as reduced fishing intensity or more good days) and also under worsening conditions (such as smaller catches or more bad days) (Chapter 7). This suggests that strict enforcement of fishing permit regulations and access to the fishery is required. However, the Federation of Fishers would need to regain the trust and confidence of fishers in order for fishers to want to be members. Fishers would need to see real benefits from the Federation, including regular meetings with fishers, monitoring and enforcement of regulations, and programs designed to help fishers in times of difficulty.

Invasive aquatic plants present near Anororo (water hyacinth, *Eichhornia crassipes*, and giant salvinia, *Salvinia molesta*) are known to degrade water quality, reducing dissolved oxygen levels in the water and potentially causing fish mortality (Howard and Harley, 1998). Anororo fishers are aware of these effects and also reported that the channels and paths through the marsh become clogged with these plants to the point where the channels are impassable, forcing fishers to either make new channels or fish elsewhere. Fishers suggested the fishery would benefit from a participatory government-implemented program to systematically remove these invasive plants from the marsh and channels. Several fishers stated that access to the marsh would be improved if these plants were removed, and fishers would no longer need to build reed walls (‘*Hamatras*’) at the lake-marsh edge to attach traps to (an activity which is illegal). However, the majority of fishers also stated that, if the plants were removed, they would change to a fishing location within the marsh (Chapter 7 – Supporting information).

Despite the perceived benefits of removing or controlling these invasive plants (e.g., improved water quality and fishing locations, increased catches and income), one caveat is that it would facilitate access into the marsh and planned strict conservation zone intended to protect habitat for the endemic lemur, and open the marsh to potential degradation. Although fishing in its purest sense does not degrade marsh habitat, other stakeholders such as marsh-product users could have a significant adverse impact on native marsh plants if they were easily accessible. Removing or controlling invasive aquatic plants is a double-edged sword and would need to be preceded by a strong education and social marketing program, as well as followed by ongoing monitoring and strict enforcement. Furthermore, the removal of invasive aquatic plants can be complex and often only a short-term solution when conducted mechanically, which may intensify the problem (i.e., water hyacinth can regenerate from broken stems). The alternative of chemical control is usually not a favourable option because of the unknown long-term effects of chemicals on the environment and surrounding communities. Although some biological control methods have been used successfully in other regions (Navarro and Phiri, 2000) and are generally the most favoured option, the risk of control agents moving on to other native aquatic plants and/or rice has not been adequately assessed. This management action is therefore only recommended with a heed of caution.

Interestingly, most of the suggestions for better management made by fishers focused on enforcement and there were overwhelmingly positive results for fishers' willingness to engage with management (Chapter 7). I believe that a locally-based monitoring team at Lake Alaotra, similar to forest monitors in the Menabe (see Sommerville et al., 2010b), would be beneficial for a number of reasons. First, some fishers appear to take advantage of the fact that the Fisheries Service is unable to monitor the entire lake and marsh effectively, and fishers can easily hide in the marsh when they hear the authorities approaching in their large underpowered boat; local monitors could fill this gap. Second, regular monitoring by peers would probably increase compliance. And third, Anororo-based fishers actually suggested this as an appropriate management action (Chapter 7).

Fishers also stated their willingness to collect information about their fishing activity and catches (Chapter 7). This type of participatory monitoring would be an effective way to further engage resource users in developing management interventions while building larger datasets than otherwise feasible, subject to challenges of entering and analysing the data. Some fishers indicated that they did not have time to participate in such a monitoring scheme, which highlights the need to keep monitoring demands on fishers to a minimum. I also believe that fishers should be compensated for their time, perhaps by providing them with free membership in the Federation of Fishers and an annual fishing permit.

The conservation planning exercise carried out with Marxan and MinPatch software provides an approach to incorporate socioeconomic data into conservation planning (Chapter 8). The results presented in Chapter 8 form a baseline from which a new spatial management plan for Lake Alaotra may be developed. A consultation process by which local stakeholders are key partners in fine-tuning the plan is now needed. Following the consultation process, a social marketing program could be the most efficient and effective way of reaching all fishers around Lake Alaotra, encouraging them to be involved and buy-in to the conservation and management plan. Engaging local communities and fostering sustainable behaviour are key components of community-based social marketing (McKenzie-Mohr, 2000) and this technique has been used successfully in other regions of Madagascar (see Bianchessi et al., 2011).

Local radio is also used widely in the region to communicate important information such as approaching cyclones and timing of the fishery closure. Radio could be used within a social marketing program to provide region-wide information about fishery regulations and their benefits at relatively low cost.

Although the recommendations suggested here are not panaceas, they would allow Durrell to take a more adaptive and participatory approach to managing activities in the Lake Alaotra wetland, while also providing additional data on which to base management decisions. Participatory approaches to management planning and ongoing monitoring will be a key factor determining the success of conservation interventions.

9.4 CONCLUSIONS

To achieve effective fisheries management and conservation outcomes, it is necessary to understand fisher perceptions and behaviour, determine the costs to fishers of management interventions, and incorporate socioeconomic factors when planning conservation actions and developing models to inform strategy. To do this it will be imperative to engage with and involve fishers as partners in management, which will (a) build trust and confidence, (b) enable fishers' interests and concerns to be taken into account, (c) provide insights about how they make fishing decisions and respond to change, and (d) support a participatory approach when identifying, implementing, and/or monitoring potential solutions. There should be a focus on short-term costs to fishers because these will have the greatest impact on the implementation and ongoing success of any conservation plan restricting fisher behaviour. Conservation planners, resource managers, researchers, and governance institutions must recognise in policy as well as practice that fishers and other resource users are agents of change and affected by change. This recognition should ensure that fishers are given a much-needed voice in management decisions and sustaining the resources and systems on which they depend.

REFERENCES

Abbot, J. I. O., and Guijt, I., 1998. Changing views on change: Participatory approaches to monitoring the environment. Discussion Paper 2. Sustainable Agriculture and Rural Livelihoods Programme, International Institute for Environment and Development, London. 98 pp.

Abernethy, K. E., 2010. Fishing for what? Understanding fisher decision-making in southwest England. PhD Thesis. University of East Anglia, Norwich. 271 pp.

Abernethy, K. E., Allison, E. H., Molloy, P. P., and Côté, I. M., 2007. Why do fishers fish where they fish? Using the ideal free distribution to understand the behaviour of artisanal reef fishers. Canadian Journal of Fisheries and Aquatic Sciences 64 (11), 1595-1604.

Adams, V. M., Mills, M., Jupiter, S. D., and Pressey, R. L., 2011. Improving social acceptability of marine protected area networks: A method for estimating opportunity costs to multiple gear types in both fished and currently unfished areas. Biological Conservation 144 (1), 350-361.

Adams, V. M., Pressey, R. L., and Naidoo, R., 2010. Opportunity costs: Who really pays for conservation? Biological Conservation 143 (2), 439-448.

Adams, W. M., 1998. Conservation and development. pp. 286-315. *In*: W. J. Sutherland (ed.), Conservation science and action. Blackwell Science, Oxford.

Adger, W. N., and Kelly, P. M., 2001. Social vulnerability and resilience. pp. 19-34. *In*: W. N. Adger, P. M. Kelly, and N. H. Ninh (eds.), Living with environmental change: Social vulnerability, adaptation and resilience in Vietnam. Routledge, London.

Akaike, H., 1974. A new look at the statistical model identification. IEEE Transactions on Automatic Control 19 (6), 716-723.

Allan, J. D., Abell, R., Hogan, Z. E. B., Revenga, C., Taylor, B. W., Welcomme, R. L., and Winemiller, K., 2005. Overfishing of inland waters. BioScience 55 (12), 1041-1051.

Allison, E. H., and Ellis, F., 2001. The livelihoods approach and management of small-scale fisheries. Marine Policy 25 (5), 377-388.

Alpert, P., 1996. Integrated conservation and development projects. BioScience 46 (11), 845-855.

Amarasinghe, U. S., and de Silva, S. S., 1999. Sri Lankan reservoir fishery: A case for introduction of a co-management strategy. Fisheries Management & Ecology 6 (5), 387-400.

Anderies, J. M., Janssen, M. A., and Ostrom, E., 2004. A framework to analyze the robustness of social-ecological systems from an institutional perspective. Ecology and Society 9 (1), 18.

Andrianandrasana, H. T., 2009. Développement des systèmes de télédétection en vue d'améliorer le suivi de l'habitat et des feux de marais au Lac Alaotra. Mémoire de Diplôme d'Etudes Approfondies en Foresterie Développement et Environnement. Université d'Antananarivo, Antananarivo. 120 pp.

Andrianandrasana, H. T., Randriamahefasoa, J., Durbin, J., Lewis, R. E., and Ratsimbazafy, J. H., 2005. Participatory ecological monitoring of the Alaotra wetlands in Madagascar. Biodiversity and Conservation 14, 2757-2774.

Ardron, J., Possingham, H. P., and Klein, C. J., editors. 2010. Marxan good practices handbook, Version 2. University of Queensland, St. Lucia, Queensland, Australia, and Pacific Marine Analysis and Research Association, Victoria, British Columbia, Canada. 165 pp.

Ariely, D., 2009. Predictably irrational, Revised ed. edition. HarperCollins, London. pp.

Arksey, H., and Knight, P., 1999. Interviewing for social scientists. Sage Publications Ltd, London. 208 pp.

Armitage, D. R., 2003. Traditional agroecological knowledge, adaptive management and the socio-politics of conservation in Central Sulawesi, Indonesia. Environmental Conservation 30 (1), 79-90.

Arponen, A., Cabeza, M., Eklund, J., Kujala, H., and Lehtomaki, J., 2010. Costs of integrating economics and conservation planning. Conservation Biology 24 (5), 1198-1204.

Aswani, S., and Hamilton, R. J., 2004a. Integrating indigenous ecological knowledge and customary sea tenure with marine and social science for conservation of bumphead parrotfish (*Bolbometopon muricatum*) in the Roviana Lagoon, Solomon Islands. Environmental Conservation 31 (01), 69-83.

Aswani, S., and Hamilton, R. J., 2004b. The value of many small vs. few large marine protected areas in the Western Solomon Islands. SPC Traditional Marine Resource Management and Knowledge Information Bulletin 16, 3-14.

Balakrishnan, P. V., Nataraajan, R., and Desai, A., 2000. Consumer rationality and economic efficiency: Is the assumed link justified? Marketing Management Journal 10 (1), 1-11.

Balirwa, J. S., Chapman, C. A., Chapman, L. J., Cowx, I. G., Geheb, K., Kaufman, L., Lowe-McConnell, R. H., Seehausen, O., Wanink, J. H., Welcomme, R. L., and Witte, F., 2003. Biodiversity and fishery sustainability in the Lake Victoria basin: An unexpected marriage? BioScience 53 (8), 703-715.

Ball, I. R., and Possingham, H. P., 2000. MARXAN (V1.8.2): Marine Reserve Design Using Spatially Explicit Annealing, a Manual, 1.8.2.

Balmford, A., and Cowling, R. M., 2006. Fusion or failure? The future of conservation biology. Conservation Biology 20 (3), 692-695.

Balmford, A., and Whitten, T., 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx 37 (2), 238-250.

Ban, N. C., 2008. Multiple perspectives for envisioning marine protected areas. PhD Thesis. Univserity of British Columbia, Vancouver. 217 pp.

Ban, N. C., Hansen, G. J. A., Jones, M., and Vincent, A. C. J., 2009. Systematic marine conservation planning in data-poor regions: Socioeconomic data is essential. Marine Policy 33 (5), 794-800.

Ban, N. C., and Klein, C. J., 2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. Conservation Letters 2 (5), 206-215.

Barbier, E. B., 2000. Valuing the environment as input: Review of applications to mangrove-fishery linkages. Ecological Economics 35 (1), 47-61.

Barbier, E. B., 2003. Habitat-fishery linkages and mangrove loss in Thailand. Contemporary Economic Policy 21 (1), 59-77.

Barrett, C. B., and Arcese, P., 1995. Are integrated conservation-development projects (ICDPs) sustainable? On the conservation of large mammals in sub-Saharan Africa. World Development 23 (7), 1073-1084.

Beger, M., Harborne, A. R., Dacles, T. P., Solandt, J.-L., and Ledesma, G. L., 2004. A framework of lessons learned from community-based marine reserves and its effectiveness in guiding a new coastal management initiative in the Philippines. Environmental Management 34 (6), 786-801.

Begossi, A., 2006. Temporal stability in fishing spots: Conservation and co-management in Brazilian artisanal coastal fisheries. Ecology and Society 11 (1), 5 [online] URL: http://www.ecologyandsociety.org/vol11/iss1/art5/.

Béné, C., 1996. Effects of market constraints, the remuneration system, and resource dynamics on the spatial distribution of fishing effort. Canadian Journal of Fisheries and Aquatic Sciences 53 (3), 563-571.

Béné, C., Belal, E., Baba, M. O., Ovie, S., Raji, A., Malasha, I., Njaya, F., Na Andi, M., Russell, A., and Neiland, A., 2009. Power struggle, dispute and alliance over local resources: Analyzing 'democratic' decentralization of natural resources through the lenses of Africa inland fisheries. World Development 37 (12), 1935-1950.

Béné, C., Macfadyen, G., and Allison, E. H., 2007. Increasing the contribution of small-scale fisheries to poverty alleviation and food security. FAO Fisheries Technical Paper No. 481, FAO, Rome. 125 pp.

Béné, C., Neiland, A., Jolley, T., Ovie, S., Sule, O., Ladu, B., Mindjimba, K., Belal, E., Tiotsop, F., Baba, M., Dara, L., Zakara, A., and Quensiere, J., 2003. Inland fisheries, poverty, and rural livelihoods in the Lake Chad Basin. Journal of Asian and African Studies 38 (1), 17-51.

Béné, C., and Tewfik, A., 2001. Fishing effort allocation and fishermen's decision making process in a multi-species small-scale fishery: Analysis of the conch and lobster fishery in Turks and Caicos Islands. Human Ecology 29 (2), 157-186.

Benstead, J. P., De Rham, P. H., Gattolliat, J.-L., Gibon, F.-M., Loiselle, P. V., Sartori, M., Sparks, J. S., and Stiassny, M. L. J., 2003. Conserving Madagascar's freshwater biodiversity. BioScience 53 (11), 1101-1111.

Berkes, F., 1985. Fishermen and 'The Tragedy of the Commons'. Environmental Conservation 12 (03), 199-206.

Berkes, F., 1990. Native subsistence fisheries: A synthesis of harvest studies in Canada. Arctic 43 (1), 35-42.

Berkes, F., 2003. Alternatives to conventional management: Lessons from small-scale fisheries. Environments 31 (1), 16.

Berkes, F., 2004. Rethinking community-based conservation. Conservation Biology 18 (3), 621-630.

Berkes, F., 2012. Implementing ecosystem-based management: Evolution or revolution? Fish and Fisheries 13 (4), 465-476.

Berkes, F., Colding, J., and Folke, C., 2000. Rediscovery of traditional ecological knowledge as adaptive management. Ecological Applications 10 (5), 1251-1262.

Berkes, F., Mahon, R., McConney, P., Pollnac, R., and Pomeroy, R. S., 2001. Managing small-scale fisheries: Alternative directions and methods. International Development Research Centre (IDRC), Ottawa. 308 pp.

Bernard, H. R., 2002. Research methods in anthropology: Qualitative and quantitative methods, 3rd edition. AltaMira Press, Walnut Creek. 753 pp.

Bianchessi, A., Andriamalala, G., Peabody, S., Harris, A., Sadowsky, B., Green, S., Cadiz, F., Canlas, C., Rojas, P., Hayashi, E., and De Wan, A., 2011. Can social marketing techniques improve compliance to marine protected areas (MPA)? A case study from Madagascar and application of lessons learned for social marketing in Philippine MPAs. Society for Conservation Biology, 25th International Congress for Conservation Biology, 5 to 9 December 2011. Auckland, New Zealand. p.

Bjarnason, T., and Thorlindsson, T., 1993. In defense of a folk model: The “skipper effect” in the Icelandic cod fishery. American Anthropologist 95 (2), 371-394.

Blueweiss, L., Fox, H., Kudzma, V., Nakashima, D., Peters, R., and Sams, S., 1978. Relationships between body size and some life history parameters. Oecologia 37 (2), 257-272.

Bockstael, N. E., and Opaluch, J. J., 1983. Discrete modelling of supply response under uncertainty: The case of the fishery. Journal of Environmental Economics and Management 10 (2), 125-137.

Bode, M., Watson, J., Iwamura, T., and Possingham, H. P., 2008a. The cost of conservation. Science 321 (5887), 340.

Bode, M., Wilson, K. A., Brooks, T. M., Turner, W. R., Mittermeier, R. A., McBride, M. F., Underwood, E. C., and Possingham, H. P., 2008b. Cost-effective global conservation spending is robust to taxonomic group. Proceedings of the National Academy of Sciences 105 (17), 6498-6501.

Bohnsack, J. A., 2000. A comparison of the short-term impact of no-take marine reserves and minimum size limits. Bulletin of Marine Science 66 (3), 635-650.

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., and White, J.-S. S., 2009. Generalized linear mixed models: A practical guide for ecology and evolution. Trends in ecology & evolution (Personal edition) 24 (3), 127-135.

Borchers, D. L., Buckland, S. T., Priede, I. G., and Ahmadi, S., 1997. Improving the precision of the daily egg production method using generalized additive models. Canadian Journal of Fisheries and Aquatic Sciences 54 (12), 2727-2742.

Branch, G. M., Hauck, M., Siqwana-Ndulo, N., and Dye, A. H., 2002. Defining fishers in the South African context: Subsistence, artisanal and small-scale commercial sectors. South African Journal of Marine Science 24 (1), 475-487.

Branch, T. A., Hilborn, R., Haynie, A. C., Fay, G., Flynn, L., Griffiths, J., Marshall, K. N., Randall, J. K., Scheuerell, J. M., Ward, E. J., and Young, M., 2006. Fleet dynamics and fishermen behavior: Lessons for fisheries managers. Canadian Journal of Fisheries and Aquatic Sciences 63 (7), 1647-1668.

Brenier, A., Ferraris, J., and Mahafina, J., 2011. Participatory assessment of the Toliara Bay reef fishery, southwest Madagascar. Madagascar Conservation and Development 6 (2), 60-67.

Briggs, C. L., 1986. Learning how to ask: A sociolinguistic appraisal of the role of the interview in social science research. Cambridge University Press, Cambridge. 155 pp.

Brondizio, E. S., Ostrom, E., and Young, O. R., 2009. Connectivity and the governance of multilevel social-ecological systems: The role of social capital. Annual Review of Environment and Resources 34 (1), 253-278.

Brown, C. J., Fulton, E. A., Possingham, H. P., and Richardson, A. J., 2012. How long can fisheries management delay action in response to ecosystem and climate change? Ecological Applications 22 (1), 298-310.

Bryman, A., 2001. Social research methods. Oxford University Press, Oxford. 540 pp.

Bunnefeld, N., Hoshino, E., and Milner-Gulland, E. J., 2011. Management strategy evaluation: A powerful tool for conservation? Trends in Ecology & Evolution 26 (9), 441-447.

Burnham, K. P., and Anderson, D. R., 2002. Model selection and multimodel inference: A practical information-theoretic approach, Second edition. Springer, New York. 488 pp.

Butterworth, D. S., and Punt, A. E., 1999. Experiences in the evaluation and implementation of management procedures. ICES Journal of Marine Science 56 (6), 985-998.

Byers, B. A., Cunliffe, R. N., and Hudak, A. T., 2001. Linking the conservation of culture and nature: A case study of sacred forests in Zimbabwe. Human Ecology 29 (2), 187-218.

Cabrera, J. L., and Defeo, O., 2001. Daily bioeconomic analysis in a multispecific artisanal fishery in Yucatan, Mexico. Aquatic Living Resources 14 (1), 19-28.

Caddy, J. F., 1999. Fisheries management in the twenty-first century: Will new paradigms apply? Reviews in Fish Biology and Fisheries 9 (1), 1-43.

Caddy, J. F., and Garibaldi, L., 2000. Apparent changes in the trophic composition of world marine harvests: The perspective from the FAO capture database. Ocean & Coastal Management 43 (8-9), 615-655.

Cameron, S. E., Williams, K. J., and Mitchell, D. K., 2008. Efficiency and concordance of alternative methods for minimizing opportunity costs in conservation planning. Conservation Biology 22 (4), 886-896.

Campbell, L. M., and Vainio-Mattila, A., 2003. Participatory development and community-based conservation: Opportunities missed for lessons learned? Human Ecology 31 (3), 417-437.

Canonico, G. C., Arthington, A., McCrary, J. K., and Thieme, M. L., 2005. The effects of introduced tilapias on native biodiversity. Aquatic Conservation: Marine and Freshwater Ecosystems 15 (5), 463-483.

Cardoso, P., Fielding, P., and Snowman, M., 2005. Overview and analysis of social, economic and fisheries information to promote artisanal fisheries management in the BCLME region - South Africa. Environmental Evaluation Unit, University of Cape Town, Cape Town, South Africa. 163 pp.

CARE, ITFC, CDC, and WCS, 2003. Reducing the costs of conservation to frontline communities in southwest Uganda. Knowledge Base Review Report. CARE

International in Uganda, Institute of Tropical Forest Conservation, Conservation Development Centre, and Wildlife Conservation Society. 130 pp.

Carwardine, J., Klein, C. J., Wilson, K. A., Pressey, R. L., and Possingham, H. P., 2009. Hitting the target and missing the point: Target-based conservation planning in context. Conservation Letters 2 (1), 4-11.

Carwardine, J., Wilson, K. A., Ceballos, G., Ehrlich, P. R., Naidoo, R., Iwamura, T., Hajkowicz, S. A., and Possingham, H. P., 2008. Cost-effective priorities for global mammal conservation. Proceedings of the National Academy of Sciences 105 (32), 11446-11450.

Carwardine, J., Wilson, K. A., Hajkowicz, S. A., Smith, R. J., Klein, C. J., Watts, M., and Possingham, H. P., 2010. Conservation planning when costs are uncertain. Conservation Biology 24 (6), 1529-1537.

Chambers, R., 2006. Participatory mapping and geographic information systems: Whose map? Who is empowered and who disempowered? Who gains and who loses? The Electronic Journal on Information Systems in Developing Countries 25 (2), 1-11.

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C., and Daily, G. C., 2006. Conservation planning for ecosystem services. Public Library of Science Biology 4 (11), e379.

Charles, A., Burbidge, C., Boyd, H., and Lavers, A., 2009. Fisheries and the marine environment in Nova Scotia: Searching for sustainability and resilience. GPI Atlantic, Saint Mary's University, Halifax, Nova Scotia. 55 pp.

Charles, A., and Wilson, L., 2009. Human dimensions of Marine Protected Areas. ICES Journal of Marine Science 66 (1), 6-15.

Charles, A. T., 1995. Fishery science: The study of fisheries systems. Aquatic Living Resources 8, 233-239.

Chen, L., Messing, I., Zhang, S., Fu, B., and Ledin, S., 2003. Land use evaluation and scenario analysis towards sustainable planning on the Loess Plateau in China--case study in a small catchment. CATENA 54 (1-2), 303-316.

Christensen, A.-S., and Raakjær, J., 2006. Fishermen's tactical and strategic decisions: A case study of Danish demersal fisheries. Fisheries Research 81 (2–3), 258-267.

Christensen, V., and Pauly, D., 2004. Placing fisheries in their ecosystem context, an introduction. Ecological Modelling 172 (2-4), 103-107.

Cinner, J., 2005. Socioeconomic factors influencing customary marine tenure in the Indo-Pacific. Ecology and Society 10 (1), 36.

Cinner, J., Fuentes, M. M. P. B., and Randriamahazo, H., 2009a. Exploring social resilience in Madagascar's marine protected areas. Ecology and Society 14 (1), 41.

Cinner, J. E., 2007. Designing marine reserves to reflect local socioeconomic conditions: Lessons from long-enduring customary management systems. Coral Reefs 26 (4), 1035-1045.

Cinner, J. E., and Bodin, Ö., 2010. Livelihood diversification in tropical coastal communities: A network-based approach to analyzing 'livelihood landscapes'. PLoS ONE 5 (8), e11999.

Cinner, J. E., Daw, T., and McClanahan, T. R., 2008. Socioeconomic factors that affect artisanal fishers' readiness to exit a declining fishery. Conservation Biology 23 (1), 124-130.

Cinner, J. E., Folke, C., Daw, T., and Hicks, C. C., 2011. Responding to change: Using scenarios to understand how socioeconomic factors may influence amplifying or dampening exploitation feedbacks among Tanzanian fishers. Global Environmental Change 21 (1), 7-12.

Cinner, J. E., Marnane, M. J., and McClanahan, T. R., 2005. Conservation and community benefits from traditional coral reef management at Ahus Island, Papua New Guinea. Conservation Biology 19 (6), 1714-1723.

Cinner, J. E., and McClanahan, T. R., 2006. Socioeconomic factors that lead to overfishing in small-scale coral reef fisheries of Papua New Guinea. Environmental Conservation 33 (1), 73-80.

Cinner, J. E., McClanahan, T. R., Graham, N. A. J., Pratchett, M. S., Wilson, S. K., and Raina, J.-B., 2009b. Gear-based fisheries management as a potential adaptive response to climate change and coral mortality. Journal of Applied Ecology 46 (3), 724-732.

Cinner, J. E., McClanahan, T. R., MacNeil, M. A., Graham, N. A. J., Daw, T. M., Mukminin, A., Feary, D. A., Rabearisoa, A. L., Wamukota, A., Jiddawi, N., Campbell, S. J., Baird, A. H., Januchowski-Hartley, F. A., Hamed, S., Lahari, R., Morove, T., and Kuange, J., 2012. Comanagement of coral reef social-ecological systems. Proceedings of the National Academy of Sciences 109 (14), 5219-5222.

Cinner, J. E., McClanahan, T. R., and Wamukota, A., 2010. Differences in livelihoods, socioeconomic characteristics, and knowledge about the sea between fishers and non-fishers living near and far from marine parks on the Kenyan coast. Marine Policy 34 (1), 22-28.

Cochrane, K. L., editor. 2002. A fishery manager's guidebook: Management measures and their application. FAO, Rome. 231 pp.

Cohen, J., 1960. A coefficient of agreement for nominal scales. Educational and Psychological Measurements 20, 37-46.

Colfer, C. J. P., Wadley, R. L., and Venkateswarlu, P., 1999. Understanding local people's use of time: A pre-condition for good co-management. Environmental Conservation 26 (1), 41-52.

Copsey, J. A., Jones, J. P. G., Andrianandrasana, H. T., Rajaonarison, L. H., and Fa, J. E., 2009a. Burning to fish: Local explanations for wetland burning in Lac Alaotra, Madagascar. Oryx 43 (3), 403-406.

Copsey, J. A., Rajaonarison, L. H., Randriamihamina, R., and Rakotoniaina, L. J., 2009b. Voices from the marsh: Livelihood concerns of fishers and rice cultivators in the Alaotra wetland. Madagascar Conservation and Development 4 (1), 25-30.

Coulthard, S., 2008. Adapting to environmental change in artisanal fisheries - Insights from a South Indian Lagoon. Global Environmental Change 18 (3), 479-489.

Coulthard, S., Johnson, D., and McGregor, J. A., 2011. Poverty, sustainability and human wellbeing: A social wellbeing approach to the global fisheries crisis. Global Environmental Change 21 (2), 453-463.

Courtenay, W. R., Williams, J. D., Britz, R., Yamamoto, M. N., and Loiselle, P. V., 2004. Identity of introduced snakeheads (Pisces, Channidae) in Hawai'i and Madagascar, with comments on ecological concerns. Bishop Museum Occassional Papers 77, 13.

Cowling, R., 2005. Maintaining the research-implementation continuum in conservation. Society for Conservation Biology Newsletter 12 (4), 1, 19.

Cowling, R. M., Egoh, B., Knight, A. T., O'Farrell, P. J., Reyers, B., Rouget, M., Roux, D. J., Welz, A., and Wilhelm-Rechman, A., 2008. An operational model for mainstreaming ecosystem services for implementation. Proceedings of the National Academy of Sciences 105 (28), 9483-9488.

Cowling, R. M., and Pressey, R. L., 2003. Introduction to systematic conservation planning in the Cape Floristic Region. Biological Conservation 112 (1-2), 1-13.

Crossman, N. D., Perry, L. M., Bryan, B. A., and Ostendorf, B., 2007. CREDOS: A Conservation Reserve Evaluation and Design Optimisation System. Environmental Modelling & Software 22 (4), 449-463.

Cucherousset, J., Paillisson, J.-M., Carpentier, A., Thoby, V., Damien, J.-P., Eybert, M.-C., Feunteun, E., and Robinet, T., 2007. Freshwater protected areas: An effective measure to reconcile conservation and exploitation of the threatened European eels (*Anguilla anguilla*)? Ecology of Freshwater Fish 16 (4), 528-538.

Daget, J., and Moreau, J., 1981. Hybridation introgressive entre deux espèces de *Sarotherodon* (Pisces, Cichlidae) dans un lac de Madagascar. Bulletin Museum National d'Histoire Naturelle Paris 4e Série 3 (A), 689-703.

Damania, R., Milner-Gulland, E. J., and Crookes, D. J., 2005. A bioeconomic analysis of bushmeat hunting. Proceedings of The Royal Society of London, Series B 272 (1560), 259-266.

Danielsen, F., Burgess, N. D., and Balmford, A., 2005. Monitoring matters: Examining the potential of locally-based approaches. Biodiversity and Conservation 14 (11), 2507-2542.

Danielsen, F., Burgess, N. D., Balmford, A., Donald, P. F., Funder, M., Jones, J. P. G., Alviola, P., Balete, D. S., Blomley, T., Brashares, J. S., Child, B., Enghoff, M., Fjeldså, J., Holt, S., Hübertz, H., Jensen, A. E., Jensen, P. M., Massao, J., Mendoza, M. M., Ngaga, Y., Poulsen, M. K., Rueda, R., Sam, M., Skielboe, T., Stuart-Hill, G., Topp-Jørgensen, E., and Yonten, D., 2009. Local participation in natural resource monitoring: A characterization of approaches. Conservation Biology 23 (1), 31-42.

Danielsen, F., Mendoza, M. M., Tagtag, A., Alviola, P. A., Balete, D. S., Jensen, A. E., Enghoff, M., and Poulsen, M. K., 2007. Increasing conservation management action by involving local people in natural resource monitoring. Ambio: A Journal of the Human Environment 36 (7), 566-570.

Davies, T. E., Beanjara, N., and Tregenza, T., 2009. A socio-economic perspective on gear-based management in an artisanal fishery in south-west Madagascar. Fisheries Management and Ecology 16 (4), 279-289.

Daw, T., Brown, K., Rosendo, S., and Pomeroy, R., 2011a. Applying the ecosystem services concept to poverty alleviation: The need to disaggregate human well-being. Environmental Conservation 38 (4), 370-379.

Daw, T., Maina, J., Cinner, J., Robinson, J., and Wamukota, A., 2011b. The spatial behaviour of artisanal fishers: Implications for fisheries management and development (Fishers in Space). Western Indian Ocean Marine Science Association (WIOMSA), Marine Science for Management (MASMA). 79 pp.

Daw, T. M., 2008. Spatial distribution of effort by artisanal fishers: Exploring economic factors affecting the lobster fisheries of the Corn Islands, Nicaragua. Fisheries Research 90 (1-3), 17-25.

Daw, T. M., Cinner, J. E., McClanahan, T. R., Brown, K., Stead, S. M., Graham, N. A. J., and Maina, J., 2012. To fish or not to fish: Factors at multiple scales affecting artisanal fishers' readiness to exit a declining fishery. PLoS ONE 7 (2), e31460.

de Silva, C. D., 1997. Genetic variation in tilapia populations in man-made reservoirs in Sri Lanka. Aquaculture International 5 (4), 339-349.

de Silva, C. D., and Ranasinghe, J., 1989. Biochemical evidence of hybrid gene introgression in some reservoir populations of tilapia in southern Sri Lanka. Aquaculture Research 20 (3), 269-277.

Dietz, T., Ostrom, E., and Stern, P. C., 2003. The struggle to govern the commons. Science 302 (5652), 1907-1912.

Dina de Pêche, 2006. Joint convention to regulate the exploitation of fish and threatened species of the Alaotra region. Durrell Wildlife Conservation Trust and Regional Department of Fishing and Fisheries Resources, Ambatondrazaka, Madagascar. 6 pp.

Dinmore, T. A., Duplisea, D. E., Rackham, B. D., Maxwell, D. L., and Jennings, S., 2003. Impact of a large-scale area closure on patterns of fishing disturbance and the consequences for benthic communities. ICES Journal of Marine Science 60 (2), 371-380.

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z.-I., Knowler, D. J., Leveque, C., Naiman, R. J., Prieur-Richard, A.-H., Soto, D., Stiassny, M. L. J., and Sullivan, C. A., 2006. Freshwater biodiversity: Importance, threats, status and conservation challenges. Biol. Rev. 81 (2), 163-182.

Durand, J.-R., 1979. Evolution des captures totales (1962-1977) et devenir des pecheries de la region du lac Tchad. Cahiers ORSTOM series Hydrobiologie 13 (1-2), 93-111.

Durbin, J., Funk, S. M., Hawkins, F., Hills, D. M., Jenkins, P. D., Moncrieff, C. B., and Ralainasolo, F. B., 2010. Investigations into the status of a new taxon of Salanoia (Mammalia: Carnivora: Eupleridae) from the marshes of Lac Alaotra, Madagascar. Systematics and Biodiversity 8 (3), 341-355.

Durbin, J., Rakotoniaina, L. J., and Randriamahefasoa, J., 2003. Project Alaotra: Using endangered species as flagships for community-based wetland conservation. pp. 1551-1555. *In*: S. M. Goodman and J. P. Benstead (eds.), The Natural History of Madagascar. University of Chicago Press, Chicago.

DWCT, and Service de Pêche, 2002. Dina de Pêche: Joint agreement to regulate the operation of fisheries and endangered species of the region of Alaotra. Ambatondrazaka, Madagascar. 11 pp.

DWCT, and Service de Pêche, 2006. Dina de Pêche: Joint agreement to regulate the operation of fisheries and endangered species of the region of Alaotra. Ambatondrazaka, Madagascar. 12 pp.

Edwards-Jones, G., 2006. Modelling farmer decision-making: Concepts, progress and challenges. Animal Science 82 (06), 783-790.

Eggert, H., and Lokina, R. B., 2010. Regulatory compliance in Lake Victoria fisheries. Environment and Development Economics 15 (2), 197-217.

Eggert, H., and Tveteras, R., 2004. Stochastic production and heterogeneous risk preferences: Commercial fishers' gear choices. American Journal of Agricultural Economics 86 (1), 199-212.

Emerton, L., 1999. Balancing the opportunity costs of wildlife conservation for communities around Lake Mburo National Park, Uganda. International Institute for Environment and Development, Evaluating Eden Series Discussion Paper No.5. pp.

Esterberg, K. G., 2002. Qualitative methods in social research. McGraw Hill, Boston. 256 pp.

Evans, L. S., Brown, K., and Allison, E. H., 2011. Factors influencing adaptive marine governance in a developing country context: A case study of Southern Kenya. Ecology and Society 16 (2), 21.

Ezealor, A. U., and Giles Jr., R. H., 1997. Vertebrate pests of a Sahelian wetland agro-ecosystem: Perceptions and attitudes of the indigenes and potential management strategies. International Journal of Pest Management 43 (2), 97-104.

FAO, 1995. Precautionary approach to fisheries. Part 1: Guidelines on the precautionary approach to capture fisheries and species introductions. Fisheries Technical Paper No. 350, Part 1, Food and Agriculture Organization, Rome. 52 pp.

FAO, 1997. Inland fisheries. FAO Technical Guidelines for Responsible Fisheries. No. 6, FAO, Rome. 36 pp.

FAO, 1999. Review of the state of world fishery resources: Inland fisheries. FAO Fisheries Circular. No. 942, FAO Inland Water Resources and Aquaculture Service, Fishery Resources Division, Rome. 53 pp.

FAO, 2003. Review of the state of world fishery resources: Inland fisheries. FAO Fisheries Circular. No. 942, Rev. 1, FAO Inland Water Resources and Aquaculture Service, Fishery Resources Division, Rome. 60 pp.

FAO, 2010. The state of world fisheries and aquaculture. FAO Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations, Rome. 218 pp.

Faraway, J. J., editor. 2006. Extending the linear model with R: Generalized linear, mixed effects and nonparametric regression models. Chapman & Hall, Taylor & Francis Group, Boca Raton, Florida. 331 pp.

FCO, 2008. Country profiles: Madagascar. Foreign and Commonwealth Office. http://www.fco.gov.uk/en/about-the-fco/country-profiles/sub-saharan-africa/madagascar. 17 April 2009.

Ferry, L., Mietton, M., Robison, L., and Erismann, J., 2009. Le lac Alaotra à Madagascar - Passé, présent et futur. Zeitschrift für Geomorphologie 53 (3), 299-318.

Finlayson, M., Bunting, S. W., Beveridge, M. C. M., Tharme, R., van Brakel, M., Atapattu, S., Coates, D., and Nguyen Khoa, S., 2012. Wetland agroecosystems. pp. 94-118. *In*: E. Boelee (ed.), Managing Agroecosystems for Sustainable Water and Food Security. CABI, Cambridge.

Fisher, R. J., 1993. Social desirability bias and the validity of indirect questioning. Journal of Consumer Research 20 (2), 303-315.

Fisher, R. J., and Tellis, G. J., 1998. Removing social desirability bias with indirect questioning: Is the cure worse than the disease? pp. 563-567. *In*: J. W. Alba and J. W. Hutchinson (eds.), Advances in Consumer Research. Association for Consumer Research, California.

Foale, S., and Macintyre, M., 2000. Dynamic and flexible aspects of land and marine tenure at West Nggela: Implications for marine resource management. Oceania 71 (1), 30-45.

Forcada, A., Valle, C., Bonhomme, P., Criquet, G., Cadiou, G., Lenfant, P., and Sanchez-Lizaso, J. L., 2009. Effects of habitat on spillover from marine protected areas to artisanal fisheries. Marine Ecology Progress Series 379, 197-211.

Foxon, T., 2006. Bounded rationality and hierarchical complexity: Two paths from Simon to ecological and evolutionary economics. Ecological Complexity 3 (4), 361-368.

Fraser, E. D. G., Dougill, A. J., Mabee, W. E., Reed, M., and McAlpine, P., 2006. Bottom up and top down: Analysis of participatory processes for sustainability indicator identification as a pathway to community empowerment and sustainable environmental management. Journal of Environmental Management 78 (2), 114-127.

Frazee, S. R., Cowling, R. M., Pressey, R. L., Turpie, J. K., and Lindenberg, N., 2003. Estimating the costs of conserving a biodiversity hotspot: A case-study of the Cape Floristic Region, South Africa. Biological Conservation 112 (1-2), 275-290.

Freckleton, R., 2011. Dealing with collinearity in behavioural and ecological data: Model averaging and the problems of measurement error. Behavioral Ecology and Sociobiology 65 (1), 91-101.

Fretwell, S. D., and Lucas, H. L., 1969. On territorial behavior and other factors influencing habitat distribution in birds. Acta Biotheoretica 19 (1), 16-36.

Froese, R., 2004. Keep it simple: Three indicators to deal with overfishing. Fish and Fisheries 5 (1), 86-91.

Froese, R., and Pauly, D., editors. 2010. FishBase, www.fishbase.org, (03/2010) edition. World Wide Web electronic publication. pp.

Fuller, R. A., McDonald-Madden, E., Wilson, K. A., Carwardine, J., Grantham, H. S., Watson, J. E. M., Klein, C. J., Green, D. C., and Possingham, H. P., 2010.

Replacing underperforming protected areas achieves better conservation outcomes. Nature 466 (7304), 365-367.

Fulton, E. A., Smith, A. D. M., Smith, D. C., and van Putten, I. E., 2011. Human behaviour: The key source of uncertainty in fisheries management. Fish and Fisheries 12 (1), 2-17.

Game, E. T., Bode, M., McDonald-Madden, E., Grantham, H. S., and Possingham, H. P., 2009. Dynamic marine protected areas can improve the resilience of coral reef systems. Ecology Letters 12 (12), 1336-1346.

Game, E. T., and Grantham, H. S., 2008. Marxan User Manual: For Marxan version 1.8.10, 1.8.10. University of Queensland, St. Lucia, Queensland, Australia, and Pacific Marine Analysis and Research Association, Victoria, British Columbia, Canada,

Game, E. T., Lipsett-Moore, G., Hamilton, R., Peterson, N., Kereseka, J., Atu, W., Watts, M., and Possingham, H., 2011. Informed opportunism for conservation planning in the Solomon Islands. Conservation Letters 4 (1), 38-46.

Garcia, G., and Goodman, S. M., 2003. Hunting of protected animals in the Parc National d'Ankarafantsika, north-western Madagascar. Oryx 37 (1), 115-118.

Gavin, M. C., Solomon, J. N., and Blank, S. G., 2010. Measuring and monitoring illegal use of natural resources. Conservation Biology 24 (1), 89-100.

Gelcich, S., Edwards-Jones, G., and Kaiser, M. J., 2005. Importance of attitudinal differences among artisanal fishers toward co-management and conservation of marine resources. Conservation Biology 19 (3), 865-875.

Gell, F. R., and Roberts, C. M., 2003. Benefits beyond boundaries: The fishery effects of marine reserves. Trends in Ecology & Evolution 18 (9), 448-455.

Gillingham, S., and Lee, P. C., 1999. The impact of wildlife-related benefits on the conservation attitudes of local people around the Selous Game Reserve, Tanzania. Environmental Conservation 26 (3), 218-228.

Gillis, D. M., 2003. Ideal free distributions in fleet dynamics: A behavioral perspective on vessel movement in fisheries analysis. Canadian Journal of Zoology 81 (2), 177-187.

Gillis, D. M., Peterman, R. M., and Tyler, A. V., 1993. Movement dynamics in a fishery: Application of the ideal free distribution to spatial allocation of effort. Canadian Journal of Fisheries and Aquatic Sciences 50 (2), 323-333.

Gintis, H., 2007. A framework for the unification of the behavioral sciences. Behavioral and Brain Sciences 30 (1), 1-16.

Gracia, A., 1996. White shrimp (*Penaeus setiferus*) recruitment overfishing. Marine and Freshwater Research 47 (1), 59-65.

Grantham, H., Agostini, V. N., Wilson, J., Mangubhai, S., Hidayat, N., Muljadi, A., Muhajir, Rotinsulu, C., Mongdong, M., Beck, M. W., and Possingham, H., 2012. A comparison of zoning analyses to inform the planning of a marine protected area network in Raja Ampat, Indonesia. Marine Policy, http://dx.doi.org/10.1016/j.marpol.2012.05.035.

Grantham, H. S., Petersen, S. L., and Possingham, H. P., 2008. Reducing bycatch in the South African pelagic longline fishery: The utility of different approaches to fisheries closures. Endangered Species Research 5, 291-299.

Greenstreet, S. P. R., Fraser, H. M., and Piet, G. J., 2009. Using MPAs to address regional-scale ecological objectives in the North Sea: Modelling the effects of fishing effort displacement. ICES Journal of Marine Science 66 (1), 90-100.

Gregg, R. E., Howard, J. H., and Snhonhiwa, F., 1998. Introgressive hybridization of tilapias in Zimbabwe. Journal of Fish Biology 52 (1), 1-10.

Grimble, R., Chan, M.-K., Aglionby, J., and Quan, J., 1995. Trees and trade-offs: A stakeholder approach to natural resource management. International Institute for Environment and Development - Sustainable Agriculture and Rural Livelihoods Programme, Gatekeeper Series No. 52. pp.

Grimes, D. I. F., Pardo-Igúzquiza, E., and Bonifacio, R., 1999. Optimal areal rainfall estimation using rainguages and satellite data. Journal of Hydrology 222, 93-108.

Guillera-Arroita, G., Lahoz-Monfort, J. J., Milner-Gulland, E. J., Young, R. P., and Nicholson, E., 2010a. Monitoring and conservation of the critically endangered Alaotran gentle lemur *Hapalemur alaotrensis*. Madagascar Conservation and Development 5 (2), 103-109.

Guillera-Arroita, G., Lahoz-Monfort, J. J., Milner-Gulland, E. J., Young, R. P., and Nicholson, E., 2010b. Using occupancy as a state variable for monitoring the critically endangered Alaotran gentle lemur. Endangered Species Research 11, 157-166.

Gunderson, L., Holling, C., and Light, S., 1995. Barriers and bridges to the renewal of ecosystems and institutions. Columbia University Press, New York. pp.

Güth, W., 2008. (Non)behavioral economics: A programmatic assessment. Zeitschrift für Psychologie / Journal of Psychology 216 (4), 244-253.

Hackel, J. D., 1999. Community conservation and the future of Africa's wildlife. Conservation Biology 13 (4), 726-734.

Halls, A. S., Arthur, R. I., Bartley, D., Felsing, M., Grainger, R., Hartmann, W., Lamberts, D., Purvis, J., Sultana, P., Thompson, P., and Walmsley, S., 2005. Guidelines for designing data collection and sharing systems for co-managed fisheries. Part 2: Technical guidelines. FAO Fisheries Technical Paper No. 494/2, FAO, Rome. 108 pp.

Halls, A. S., Paxton, B. R., Hall, N., Pengbun, N., Lieng, S., Pengby, N., and So, N., 2011. The stationary trawl (Dai) fishery of the Tonle Sap-Great Lake, Cambodia. Mekong River Commission, Vientiane. pp.

Halls, A. S., and Welcomme, R. L., 2004. Dynamics of river fish populations in response to hydrological conditions: A simulation study. River Research and Applications 20 (8), 985-1000.

Halpern, B. S., 2003. The impact of marine reserves: Do reserves work and does reserve size matter? Ecological Applications 13 (1), S117-S137.

Halpern, B. S., Regan, H. M., Possingham, H. P., and McCarthy, M. A., 2006. Accounting for uncertainty in marine reserve design. Ecology Letters 9 (1), 2-11.

Halpern, B. S., and Warner, R. R., 2002. Marine reserves have rapid and lasting effects. Ecology Letters 5, 361-366.

Hamerlynck, O., Duvail, S., Vandepitte, L., Kindinda, K., Nyingi, D. W., Paul, J.-L., Yanda, P. Z., Mwakalinga, A. B., Mgaya, Y. D., and Snoeks, J., 2011. To connect or not to connect - floods, fisheries and livelihoods in the Lower Rufiji floodplain lakes, Tanzania. Hydrological Sciences Journal 56 (8), 1436-1451.

Hardin, G., 1968. The tragedy of the commons. Science 162 (3859), 1243-1248.

Hart, P. J. B., and Pitcher, T. J., 1998. Conflict, consent and cooperation: The utility of an evolutionary perspective on individual human behaviour in fisheries management. pp. 215-225. *In*: T. J. Pitcher, D. Pauly, and P. Hart (eds.), Reinventing fisheries management. Kluwer Academic Publishers, Dordrecht, Netherlands.

Hastie, R., and Dawes, R. M., 2010. Rational choice in an uncertain world: The psychology of judgement and decision making, 2nd ed. edition. Sage Publications, Thousand Oaks. pp.

Hauck, M., 2000. Draft Report 6: Review of literature on artisanal and subsistence fishers. Subsistence Fisheries Task Group (SFTG), Institute of Criminology: Social Justice Resource Project, Faculty of Law, University of Cape Town. pp.

Hauser, C. E., Pople, A. R., and Possingham, H. P., 2006. Should managed populations be monitored every year? Ecological Applications 16 (2), 807-819.

Hawkins, F., Andriamasimanana, R., The Seing, S., and Rabeony, Z., 2000. The sad story of Alaotra little grebe *Tachybaptus rufolavatus*. Bulletin African Bird Club 7 (2), 115-117.

Hawkins, J. P., and Roberts, C. M., 2004. Effects of artisanal fishing on caribbean coral reefs. Conservation Biology 18 (1), 215-226.

Heemskerk, M., Wilson, K., and Pavao-Zuckerman, M., 2003. Conceptual models as tools for communication across disciplines. Conservation Ecology 7 (3), 8-20.

Heinz, S. K., Conradt, L., Wissel, C., and Frank, K., 2005. Dispersal behaviour in fragmented landscapes: Deriving a practical formula for patch accessibility. Landscape Ecology 20, 83-99.

Helser, T. E., Punt, A. E., and Methot, R. D., 2004. A generalized linear mixed model analysis of a multi-vessel fishery resource survey. Fisheries Research 70 (2–3), 251-264.

Hiddink, J. G., Hutton, T., Jennings, S., and Kaiser, M. J., 2006. Predicting the effects of area closures and fishing effort restrictions on the production, biomass, and species richness of benthic invertebrate communities. ICES Journal of Marine Science: Journal du Conseil 63 (5), 822-830.

Hilborn, R., 1985. Fleet dynamics and individual variation: Why some people catch more fish than others. Canadian Journal of Fisheries and Aquatic Sciences 42 (1), 2-13.

Hilborn, R., 2007. Managing fisheries is managing people: What has been learned? Fish and Fisheries 8 (4), 285-296.

Hilborn, R., Branch, T. A., Ernst, B., Magnusson, A., Minte-Vera, C. V., Scheuerell, M. D., and Valero, J. L., 2003. State of the world's fisheries. Annual Review of Environment and Resources 28, 15.1-15.40.

Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L. W., Mangel, M., Orensanz, J., Parma, A., Rice, J., Bell, J., Cochrane, K. L., Garcia, S., Hall, S. J., Kirkwood, G. P., Sainsbury, K., Stefansson, G., and Walters, C., 2004. When can marine reserves improve fisheries management? Ocean & Coastal Management 47 (3-4), 197-205.

Hilborn, R., and Walters, C. J., 1992. Quantitative fisheries stock assessment: Choice, dynamics, and uncertainty. Chapman & Hall, London. pp.

Hill, C. M., Osborn, F. V., and Plumptre, A. J., 2002. Human-wildlife conflict: Identifying the problem and possible solutions. Albertine Rift Technical Report Series No. 1. Wildlife Conservation Society, 140.

Hill, N. A. O., 2011. Livelihood diversification for conservation: Interactions between seaweed farming and fishing in Danajon Bank, central Philippines. PhD Thesis. Imperial College London & Institute of Zoology, London. 215 pp.

Hinz, H., Murray, L. G., Lambert, G. I., Hiddink, J. G., and Kaiser, M. J., 2012. Confidentiality over fishing effort data threatens science and management progress. Fish and Fisheries, doi: 10.1111/j.1467-2979.2012.00475.x.

Hoang Fagerström, M. H., Messing, I., and Wen, Z. M., 2003. A participatory approach for integrated conservation planning in a small catchment in Loess Plateau, China: Part I. Approach and Methods. CATENA 54 (1-2), 255-269.

Hockley, N. J., Jones, J. P. G., Andriahajaina, F. B., Manica, A., Ranambitsoa, E. H., and Randriamboahary, J. A., 2005. When should communities and conservationists monitor exploited resources? Biodiversity and Conservation 14 (11), 2795-2806.

Holland, D. S., 2008. Are fishermen rational? A fishing expedition. Marine Resource Economics 23, 325-344.

Holland, D. S., and Herrera, G. E., 2009. Uncertainty in the management of fisheries: Contradictory implications and a new approach. Marine Resource Economics 24, 289-299.

Holland, D. S., and Sutinen, J. G., 1999. An empirical model of fleet dynamics in New England trawl fisheries. Canadian Journal of Fisheries and Aquatic Sciences 56 (2), 253-264.

Holland, D. S., and Sutinen, J. G., 2000. Location choice in New England trawl fisheries: Old habits die hard. Land Economics 76 (1), 133-149.

Holling, C. S., Berkes, F., and Folke, C., 1998. Science, sustainability and resource management. pp. 342-362. *In*: F. Berkes and C. Folke (eds.), Linking social and ecological systems: Management practices and social mechanisms for building resilience. Cambridge University Press, Cambridge.

Holmes, C. M., 2003. The influence of protected area outreach on conservation attitudes and resource use patterns: A case study from western Tanzania. Oryx 37 (3), 305-315.

Honneland, G., 1999. A model of compliance in fisheries: Theoretical foundations and practical application. Ocean and Coastal Management 42, 699-716.

Howard, G. W., and Harley, K. L. S., 1998. How do floating aquatic weeds affect wetland conservation and development? How can these effects be minimised? Wetlands Ecology and Management 5 (3), 215-225.

Hudson, M., 2011. Estimating population trends in elusive species using dynamic occupancy modelling; the Critically Endangered Alaotran gentle lemur. Master of Science Thesis. Imperial College London, London. 82 pp.

Hulme, D., and Murphree, M. W., editors. 2001. African wildlife and livelihoods: The promise and performance of community conservation. James Curry, Oxford. 336 pp.

Iachetti, P., 2007. A decision-support framework for conservation planning in the central interior ecoregion of British Columbia, Canada. Nature Conservancy of Canada, Alcoa Foundation Conservation and Sustainability Fellowship, and World Conservation Union (IUCN). 113 pp.

Ikiara, M. M., and Odink, J., 2000. Fishermen resistance to exit fisheries. Marine Resource Economics 14, 199-213.

Inamdar, A., Jode, H. d., Lindsay, K., and Cobb, S., 1999. Capitalizing on nature: Protected area management. Science 283 (5409), 1856-1857.

Infield, M., 1988. Attitudes of a rural community towards conservation and a local conservation area in Natal, South Africa. Biological Conservation 45 (1), 21-46.

Infield, M., and Namara, A., 2001. Community attitudes and behaviour towards conservation: An assessment of a community conservation programme around Lake Mburo National Park, Uganda. Oryx 35 (1), 48-60.

Islam, M., and Braden, J. B., 2006. Bio-economic development of floodplains: Farming versus fishing in Bangladesh. Environment and Development Economics 11 (1), 95-126.

IUCN, 2003. Recommendations of the Vth IUCN World Parks Congress. IUCN, Gland, Switzerland. 80 pp.

IUCN, 2010. IUCN Red List of Threatened Species. Version 2010.2. IUCN Species Survival Commission. www.iucnredlist.org.

Jacobsen, R. B., Wilson, D. C. K., and Ramirez-Monsalve, P., 2012. Empowerment and regulation – dilemmas in participatory fisheries science. Fish and Fisheries 13 (3), 291-302.

Jacquet, J., and Pauly, D., 2008. Funding priorities: Big barriers to small-scale fisheries. Conservation Biology 22 (4), 832-835.

Janis, I. L., 1982. Groupthink: Psychological studies of policy decisions and fiascoes. Houghton-Mifflin, Boston, MA. pp.

Jarosz, L. A., 1994. Taboo and time-work experience in Madagascar. Geographical Review 84 (4), 439-450.

Jennings, S., 2009. The role of marine protected areas in environmental management. ICES Journal of Marine Science 66 (1), 16-21.

Jentoft, S., 2000. Legitimacy and disappointment in fisheries management. Marine Policy 24 (2), 141-148.

Jentoft, S., McCay, B., and Wilson, D., 1998. Social theory and fisheries co-management. Marine Policy 22, 423-436.

Johannes, R. E., 1978. Traditional marine conservation methods in Oceania and their demise. Annual Review of Ecology and Systematics 9 (1), 349-364.

Johannes, R. E., 1998. Government-supported, village-based management of marine resources in Vanuatu. Ocean & Coastal Management 40 (2-3), 165-186.

Johannesen, A. B., 2006. Designing integrated conservation and development projects (ICDPs): Illegal hunting, wildlife conservation, and the welfare of the local people. Environment and Development Economics 11, 247-267.

Johannesen, A. B., 2007. Protected areas, wildlife conservation, and local welfare. Ecological Economics 62 (1), 126-135.

Jones, B. D., 1999. Bounded rationality. Annual Review of Political Science 2 (1), 297-321.

Jones, J. P. G., Andriamarovololona, M. M., Hockley, N., Gibbons, J. M., and Milner-Gulland, E. J., 2008. Testing the use of interviews as a tool for monitoring trends in the harvesting of wild species. Journal of Applied Ecology 45 (4), 1205-1212.

Jones, J. P. G., Rasamy, J. R., Harvey, A., Toon, A., Oidtmann, B., Randrianarison, M. H., Raminosoa, N., and Ravoahangimalala, O. R., 2009. The perfect invader: A parthenogenic crayfish poses a new threat to Madagascar's freshwater biodiversity. Biological Invasions 11 (6), 1475-1482.

Julien, M. H., Griffith, M. W., and Stanley, J. N., editors. 2001. Biological control of water hyacinth 2. Australian Centre for International Agricultural Research (ACIAR) Monograph No. 79, Canberra. 91 pp.

Jupiter, S. D., and Egli, D. P., 2011. Ecosystem-based management in Fiji: Successes and challenges after five years of implementation. Journal of Marine Biology 2011 (Article ID 940765), 14.

Kacelnik, A., Krebs, J. R., and Bernstein, C., 1992. The ideal free distribution and predator-prey populations. Trends in Ecology & Evolution 7 (2), 50-55.

Kahneman, D., 2003. Maps of bounded rationality: Psychology for behavioral economics. The American Economic Review 93 (5), 1449-1475.

Kahneman, D., 2011. Thinking, fast and slow. Allen Lane, London. pp.

Kahneman, D., and Tversky, A., 1979. Prospect theory: An analysis of decision under risk. Econometrica 47 (2), 263-291.

Keane, A., Jones, J. P. G., Edwards-Jones, G., and Milner-Gulland, E. J., 2008. The sleeping policeman: Understanding issues of enforcement and compliance in conservation. Animal Conservation 11 (2), 75-82.

Keane, A., Ramarolahy, A. A., Jones, J. P. G., and Milner-Gulland, E. J., 2011. Evidence for the effects of environmental engagement and education on knowledge of wildlife laws in Madagascar. Conservation Letters 4 (1), 55-63.

Kell, L. T., Pilling, G. M., Kirkwood, G. P., Pastoors, M., Mesnil, B., Korsbrekke, K., Abaunza, P., Aps, R., Biseau, A., Kunzlik, P., Needle, C., Roel, B. A., and Ulrich-Rescan, C., 2005. An evaluation of the implicit management procedure used for some ICES roundfish stocks. ICES Journal of Marine Science 62 (4), 750-759.

Kellner, J. B., Tetreault, I., Gaines, S. D., and Nisbet, R. M., 2007. Fishing the line near marine reserves in single and multispecies fisheries. Ecological Applications 17 (4), 1039-1054.

Kiener, A., 1962. Evolution de la pisciculture et de la pêche dans les eaux intérieures de Madagascar depuis 1950. Bulletin de Madagascar 199, 1033-1044.

King, M. F., and Bruner, G. C., 2000. Social desirability bias: A neglected aspect of validity testing. Psychology and Marketing 17 (2), 79-103.

Klein, C. J., Ban, N. C., Halpern, B. S., Beger, M., Game, E. T., Grantham, H. S., Green, A., Klein, T. J., Kininmonth, S., Treml, E., Wilson, K., and Possingham, H. P., 2010. Prioritizing land and sea conservation investments to protect coral reefs. PLoS ONE 5 (8), e12431.

Klein, C. J., Wilson, K. A., Watts, M., Stein, J., Carwardine, J., Mackey, B., and Possingham, H. P., 2009. Spatial conservation prioritization inclusive of wilderness quality: A case study of Australia's biodiversity. Biological Conservation 142 (7), 1282-1290.

Knight, A. T., Cowling, R. M., and Campbell, B. M., 2006a. An operational model for implementing conservation action. Conservation Biology 20 (2), 408-419.

Knight, A. T., Cowling, R. M., Rouget, M., Balmford, A., Lombard, A. T., and Campbell, B. M., 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. Conservation Biology 22 (3), 610-617.

Knight, A. T., Driver, A., Cowling, R. M., Maze, K., Desmet, P. G., Lombard, A. T., Rouget, M., Botha, M. A., Boshoff, A. F., Castley, J. G., Goodman, P. S., Mackinnon, K., Pierce, S. M., Sims-Castley, R., Stewart, W. I., and von Hase, A., 2006b. Designing systematic conservation assessments that promote effective implementation: Best practice from South Africa. Conservation Biology 20 (3), 739-750.

Kolding, J., and van Zwieten, P., 2012. Relative lake level fluctuations and their influence on productivity and resilience in tropical lakes and reservoirs. Fisheries Research 115-116, 99-109.

Kotler, P., and Zaltman, G., 1971. Social marketing: An approach to planned social change. Journal of Marketing 35 (3), 3-12.

Kremen, C., Cameron, A., Moilanen, A., Phillips, S. J., Thomas, C. D., Beentje, H., Dransfield, J., Fisher, B. L., Glaw, F., Good, T. C., Harper, G. J., Hijmans, R. J., Lees, D. C., Louis, E., Jr., Nussbaum, R. A., Raxworthy, C. J., Razafimpahanana, A., Schatz, G. E., Vences, M., Vieites, D. R., Wright, P. C., and Zjhra, M. L., 2008. Aligning conservation priorities across taxa in Madagascar with high-resolution planning tools. Science 320 (5873), 222-226.

Kremen, C., Colwell, R. K., Erwin, T. L., Murphy, D. D., Noss, R. F., and Sanjayan, M. A., 1993. Terrestrial arthropod assemblages: Their use in conservation planning. Conservation Biology 7 (4), 796-808.

Kremen, C., Raymond, I., and Lance, K., 1998. An interdisciplinary tool for monitoring conservation impacts in Madagascar. Conservation Biology 12 (3), 549-563.

Krueger, R. A., and Casey, M. A., 2000. Focus groups: A practical guide for applied research, 3rd edition edition. Sage Publications Inc., Thousand Oaks. 213 pp.

Lahoz-Monfort, J. J., Guillera-Arroita, G., Milner-Gulland, E. J., Young, R. P., and Nicholson, E., 2010. Satellite imagery as a single source of predictor variables for habitat suitability modelling: How Landsat can inform the conservation of a critically endangered lemur. Journal of Applied Ecology 47 (5), 1094-1102.

Landis, J. R., and Koch, G. G., 1977. The measurement of observer agreement for categorical data. Biometrics 33 (1), 159-174.

Le Courtois, S., 2010. Household choices in rice cultivation in a social-ecological system and impacts on productivity: Lessons from Anororo, Lac Alaotra, Madagascar. Master of Science Thesis. Imperial College London, London. 79 pp.

Leal, D., 1998. Community-run fisheries: Avoiding the "Tragedy of the Commons". Population and Environment 19 (3), 225-245.

Lee, K. N., 1999. Appraising adaptive management. Conservation Ecology 3 (2), 3 [online] URL: http://www.consecol.org/vol3/iss2/art3/.

Levin, P. S., Kaplan, I., Grober-Dunsmore, R., Chittaro, P. M., Oyamada, S., Andrews, K., and Mangel, M., 2009. A framework for assessing the biodiversity and fishery aspects of marine reserves. Journal of Applied Ecology 46 (4), 735-742.

Lewis, D. M., and Phiri, A., 1998. Wildlife snaring - an indicator of community response to a community-based conservation project. Oryx 32 (2), 111-121.

Linke, S., Kennard, M. J., Hermoso, V., Olden, J. D., Stein, J., and Pusey, B. J., 2012. Merging connectivity rules and large-scale condition assessment improves conservation adequacy in river systems. Journal of Applied Ecology, doi: 10.1111/j.1365-2664.2012.02177.x.

Linke, S., Turak, E., and Nel, J., 2011. Freshwater conservation planning: The case for systematic approaches. Freshwater Biology 56 (1), 6-20.

Lorenzen, K., 1996. A simple von Bertalanffy model for density-dependent growth in extensive aquaculture, with an application to common carp (*Cyprinus carpio*). Aquaculture 142, 191-205.

Lorenzen, K., 2005. Population dynamics and potential of fisheries stock enhancement: Practical theory for assessment and policy analysis. Philosophical Transactions of The Royal Society B 360, 171-189.

Lorenzen, K., 2008. Understanding and managing enhancement fisheries systems. Reviews in Fisheries Science 16 (1-3), 10-23.

Lorenzen, K., Almeida, O., Arthur, R., Garaway, C., and Nguyen Khoa, S., 2006. Aggregated yield and fishing effort in multispecies fisheries: An empirical analysis. Canadian Journal of Fisheries and Aquatic Sciences 63, 1334-1343.

Lucas Jr., R. E., 1972. Expectations and the neutrality of money. Journal of Economic Theory 4 (2), 103-124.

Ludwig, D., Hilborn, R., and Waters, C., 1993. Uncertainty, resource exploitation, and conservation: Lessons from history. Science 260 (5104), 17-36.

Lunn, K. E., and Dearden, P., 2006. Monitoring small-scale marine fisheries: An example from Thailand's Ko Chang archipelago. Fisheries Research 77 (1), 60-71.

Maccord, P., Silvano, R., Ramires, M., Clauzet, M., and Begossi, A., 2007. Dynamics of artisanal fisheries in two Brazilian Amazonian reserves: Implications to co-management. Hydrobiologia 583 (1), 365-376.

Maiorano, L., Bartolino, V., Colloca, F., Abella, A., Belluscio, A., Carpentieri, P., Criscoli, A., Jona Lasinio, G., Mannini, A., Pranovi, F., Reale, B., Relini, G., Viva, C., and Ardizzone, G. D., 2009. Systematic conservation planning in the Mediterranean: A flexible tool for the identification of no-take marine protected areas. ICES Journal of Marine Science 66 (1), 137-146.

Mapstone, B. D., Little, L. R., Punt, A. E., Davies, C. R., Smith, A. D. M., Pantus, F., McDonald, A. D., Williams, A. J., and Jones, A., 2008. Management strategy evaluation for line fishing in the Great Barrier Reef: Balancing conservation and multi-sector fishery objectives. Fisheries Research 94 (3), 315-329.

Margules, C. R., and Pressey, R. L., 2000. Systematic conservation planning. Nature 405 (6783), 243-253.

Margules, C. R., and Sarkar, S., 2007. Systematic Conservation Planning. Cambridge University Press, New York. 278 pp.

Marine Resources Assessment Group, 2009. Fisheries and governance. Policy Brief 5 Fisheries Management Science Programme (FMSP), London, UK. 4 pp.

Martin-Smith, K. M., Samoilys, M. A., Meeuwig, J. J., and Vincent, A. C. J., 2004. Collaborative development of management options for an artisanal fishery for seahorses in the central Philippines. Ocean & Coastal Management 47 (3-4), 165-193.

Mascia, M. B., Claus, C. A., and Naidoo, R., 2010. Impacts of marine protected areas on fishing communities. Conservation Biology 24 (5), 1424-1429.

Matsuishi, T., Muhoozi, L., Mkumbo, O., Budeba, Y., Njiru, M., Asila, A., Othina, A., and Cowx, I. G., 2006. Are the exploitation pressures on the Nile perch fisheries resources of Lake Victoria a cause for concern? Fisheries Management & Ecology 13 (1), 53-71.

McClanahan, T., Davies, J., and Maina, J., 2005. Factors influencing resource users and managers' perceptions towards marine protected area management in Kenya. Environmental Conservation 32 (1), 42-49.

McClanahan, T. R., 2007. Management of area and gear in Kenyan coral reefs. pp. 166-185. *In*: T. R. McClanahan and J. C. Castilla (eds.), Fisheries management: Progress towards sustainability. Blackwell Press, London.

McClanahan, T. R., 2010. Effects of fisheries closures and gear restrictions on fishing income in a Kenyan coral reef. Conservation Biology 24 (6), 1519-1528.

McClanahan, T. R., and Castilla, J. C., editors. 2007. Fisheries management: Progress towards sustainability. Blackwell Press, London. pp.

McClanahan, T. R., Cinner, J. E., Maina, J., Graham, N. A. J., Daw, T. M., Stead, S. M., Wamukota, A., Brown, K., Ateweberhan, M., Venus, V., and Polunin, N. V. C., 2008a. Conservation action in a changing climate. Conservation Letters 1, 53-59.

McClanahan, T. R., Graham, N. A. J., Wilson, S. K., Letourneur, Y., and Fisher, R., 2009. Effects of fisheries closure size, age, and history of compliance on coral reef fish communities in the western Indian Ocean. Marine Ecology Progress Series 396, 99-109.

McClanahan, T. R., Hicks, C. C., and Darling, E. S., 2008b. Malthusian overfishing and efforts to overcome it on Kenyan coral reefs. Ecological Applications 18 (6), 1516-1529.

McClanahan, T. R., and Mangi, S. C., 2004. Gear-based management of a tropical artisanal fishery based on species selectivity and capture size. Fisheries Management & Ecology 11 (1), 51-60.

McCook, L. J., Ayling, T., Cappo, M., Choat, J. H., Evans, R. D., De Freitas, D. M., Heupel, M., Hughes, T. P., Jones, G. P., Mapstone, B., Marsh, H., Mills, M., Molloy, F. J., Pitcher, C. R., Pressey, R. L., Russ, G. R., Sutton, S., Sweatman, H., Tobin, R., Wachenfeld, D. R., and Williamson, D. H., 2010. Adaptive management of the Great Barrier Reef: A globally significant demonstration of the benefits of networks of marine reserves. Proceedings of the National Academy of Sciences 107 (43), 18278-18285.

McKenzie-Mohr, D., 2000. New ways to promote proenvironmental behavior: Promoting sustainable behavior: An introduction to community-based social marketing. Journal of Social Issues 56 (3), 543-554.

McKenzie-Mohr, D., and Smith, W., 1999. Fostering sustainable behavior: An introduction to community-based social marketing. New Society Publishers, Gabriola Island, British Columbia, Canada. 160 pp.

McNeely, J. A., 1993. Parks for life: Report of the IVth World Congress on National Parks and Protected Areas. IUCN, Gland, Switzerland. 260 pp.

Milfont, T., 2009. The effects of social desirability on self-reported environmental attitudes and ecological behaviour. The Environmentalist 29 (3), 263-269.

Milligan, H., 2009. Modelling the distributions of wetland birds of Madagascar: Implications for conservation and management. Master of Science Thesis. University College London, London. 93 pp.

Mills, M., Pressey, R. L., Weeks, R., Foale, S., and Ban, N. C., 2010. A mismatch of scales: Challenges in planning for implementation of marine protected areas in the Coral Triangle. Conservation Letters 3 (5), 291-303.

Milner-Gulland, E. J., 2012. Interactions between human behaviour and ecological systems. Philosophical Transactions of the Royal Society B: Biological Sciences 367 (1586), 270-278.

Milner-Gulland, E. J., and Rowcliffe, J. M., 2007. Conservation and sustainable use: A handbook of techniques. Oxford University Press, Oxford. 310 pp.

Minten, B., Dorosh, P., Daba, M.-H., Jenn-Treyer, O., Magnay, J., and Razafintsalama, Z., 2006. Rice markets in Madagascar in disarray: Policy options for increased efficiency and price stabilization (Working Paper No.101). World Bank. pp.

Mölsä, H., Reynolds, J. E., Coenen, E. J., and Lindqvist, O. V., 1999. Fisheries research towards resource management on Lake Tanganyika. Hydrobiologia 407 (0), 1-24.

Moore, J., Balmford, A., Allnutt, T., and Burgess, N., 2004. Integrating costs into conservation planning across Africa. Biological Conservation 117 (3), 343-350.

Moreau, J., 1979a. Essai d'application au Lac Alaotra (Madagascar) d'un modèle d'étude des pêcheries pour les plaines d'inondation intertropicales. Cahiers ORSTOM series Hydrobiologie 13 (1-2), 83-91.

Moreau, J., 1979b. Le Lac Alaotra a Madagascar: Cinquante ans d'aménagement des pêches. Cahiers ORSTOM series Hydrobiologie 13 (3-4), 171-179.

Moreau, J., 1982. Etude du cycle reproducteur de *Tilapia rendalli* et *Sarotherodon macrochir* dans un lac tropical d'altitude: le lac Alaotra (Madagascar). Acta Oecologica 3 (1), 3-22.

Moreau, J., and Moreau, I., 1987. Fitting of von Bertalanffy growth function (VBGF) with two growth checks per year. Journal of Applied Ichthyology 3, 56-60.

Morse, J. R., 1997. Who is rational economic man? Social Philosophy and Policy 14 (1), 179-206.

Moustakas, A., Silvert, W., and Dimitromanolakis, A., 2006. A spatially explicit learning model of migratory fish and fishers for evaluating closed areas. Ecological Modelling 192 (1-2), 245-258.

Mugisha, J. Y. T., and Ddumba, H., 2007. The dynamics of a fisheries model with feeding patterns and harvesting: *Lates niloticus* and *Oreochromis niloticus* in Lake Victoria. Applied Mathematics and Computation 186, 142-158.

Muller, J., and Albers, H. J., 2004. Enforcement, payments, and development projects near protected areas: How the market setting determines what works where. Resource and Energy Economics 26, 185-204.

Munro, J. L., editor. 1983. Caribbean coral reef fishery resources. International Center for Living Aquatic Resources Management, Manila, Philippines. 276 pp.

Murdoch, W., Polasky, S., Wilson, K. A., Possingham, H. P., Kareiva, P., and Shaw, R., 2007. Maximizing return on investment in conservation. Biological Conservation 139 (3-4), 375-388.

Musick, J. A., and Bonfil, R., editors. 2005. Management techniques for elasmobranch fisheries. Food and Agriculture Organization of the United Nations, Rome. 251 pp.

Muth, J. F., 1961. Rational expectations and the theory of price movements. Econometrica 29 (3), 315-335.

Mutschler, T., 2002. Alaotran gentle lemur: Some aspects of its behavioral ecology. Evolutionary Anthropology 11 (S1), 101-104.

Mutschler, T., 2003. Lac Alaotra. pp. 1530-1534. *In*: S. M. Goodman and J. P. Benstead (eds.), The Natural History of Madagascar. University of Chicago Press, Chicago.

Mutschler, T., and Feistner, A. T. C., 1995. Conservation status and distribution of the Alaotran gentle lemur *Hapalemur griseus alaotrensis*. Oryx 29 (04), 267-274.

Mutschler, T., Feistner, A. T. C., and Nievergelt, C. M., 1998. Preliminary field data on group size, diet and activity in the Alaotran Gentle Lemur *Hapalemur griseus alaotrensis*. Folia Primatologica 69 (5), 325-330.

Mutschler, T., Nievergelt, C. M., and Feistner, A. T. C., 1994. Biology and conservation of *Hapalemur griseus alaotrensis*. Jersey Wildlife Preservation Trust, Jersey. 40 pp.

Mutschler, T., Randrianarisoa, A. J., and Feistner, A. T. C., 2001. Population status of the Alaotran gentle lemur *Hapalemur griseus alaotrensis*. Oryx 35 (2), 152-157.

Mutschler, T., and Tan, C. L., 2003. *Hapalemur*, bamboo or gentle lemurs. pp. 1324-1329. *In*: S. M. Goodman and J. P. Benstead (eds.), The Natural History of Madagascar. University of Chicago Press, Chicago.

Naidoo, R., and Adamowicz, W. L., 2006. Modeling opportunity costs of conservation in transitional landscapes. Conservation Biology 20 (2), 490-500.

Naidoo, R., Balmford, A., Ferraro, P. J., Polasky, S., Ricketts, T. H., and Rouget, M., 2006. Integrating economic costs into conservation planning. Trends in Ecology & Evolution 21 (12), 681-687.

Navarro, L., and Phiri, G., editors. 2000. Water hyacinth in Africa and the Middle East: A survey of problems and solutions. International Development Research Centre, Ottawa. 119 pp.

Needle, C. L., 2008. Management strategy evaluation for North Sea haddock. Fisheries Research 94 (2), 141-150.

Neiland, A. E., Bennett, E., and Townsley, P., 2005. Participatory research approaches - what have we learned? The experience of the DFID Renewable Natural Resources Research Strategy (RNRRS) Programme 1995-2005. 7 pp.

Nel, J. L., Reyers, B., Roux, D. J., Dean Impson, N., and Cowling, R. M., 2011. Designing a conservation area network that supports the representation and persistence of freshwater biodiversity. Freshwater Biology 56 (1), 106-124.

Newmark, W. D., Leonard, N. L., Sariko, H. I., and Gamassa, D.-G. M., 1993. Conservation attitudes of local people living adjacent to five protected areas in Tanzania. Biological Conservation 63 (2), 177-183.

Newmark, W. D., Manyanza, D. N., Gamassa, D.-G. M., and Sariko, H. I., 1994. The conflict between wildlife and local people living adjacent to protected areas in Tanzania: Human density as a predictor. Conservation Biology 8 (1), 249-255.

Nievergelt, C. M., Mutschler, T., Feistner, A. T. C., and Woodruff, D. S., 2002. Social system of the Alaotran gentle lemur (*Hapalemur griseus alaotrensis*): Genetic characterization of group composition and mating system. American Journal of Primatology 57 (4), 157-176.

NRCS, 2009. Conservation planning and application. United States Department of Agriculture, Natural Resources Conservation Service. http://www.nh.nrcs.usda.gov/technical/ConservationPlanning/ConservationPlanning.html. 9 April 2009.

Nyhus, P., Westley, F. R., Lacy, R. C., and Miller, P. S., 2002. A role for natural resource social science in biodiversity risk assessment. Society and Natural Resources 15 (10), 923-932.

Obura, D. O., 2001. Participatory monitoring of shallow tropical marine fisheries by artisanal fishers in Diani, Kenya. Bulletin of Marine Science 69 (2), 777-791.

Oliver, J. D., 1993. A review of the biology of giant salvinia (*Salvinia molesta* Mitchell). Journal of Aquatic Plant Management 31, 227-231.

Ormsby, A., and Kaplin, B. A., 2005. A framework for understanding community resident perceptions of Masoala National Park, Madagascar. Environmental Conservation 32 (2), 156-164.

Ostrom, E., 2007a. A diagnostic approach for going beyond panaceas. Proceedings of the National Academy of Sciences 104 (39), 15181-15187.

Ostrom, E., 2007b. The governance challenge: Matching institutions to the structure of social-ecological systems. *In*: S. Levin (ed.), The Princeton Guide to Ecology. Princeton University Press, Princeton, NJ.

Ostrom, E., 2009. A general framework for analyzing sustainability of social-ecological systems. Science 325 (5939), 419-422.

Pálsson, G., and Durrenberger, E. P., 1990. Systems of production and social discourse: The skipper effect revisited. American Anthropologist 92 (1), 130-141.

Parker, G. A., and Sutherland, W. J., 1986. Ideal free distributions when individuals differ in competitive ability: Phenotype-limited ideal free models. Animal Behaviour 34 (4), 1222-1242.

Parma, A. M., and NCEAS Working Group on Population Management, 1998. What can adaptive management do for our fish, forests, food, and biodiversity? Integrative Biology: Issues, News, and Reviews 1 (1), 16-26.

Pauly, D., Christensen, V., Dalsgaard, A. J. T., Froese, R., and Torres, F., Jr., 1998. Fishing down marine food webs. Science 279 (5352), 860-863.

Pauly, D., Christensen, V., Guenette, S., Pitcher, T. J., Sumaila, U. R., Walters, C. J., Watson, R., and Zeller, D., 2002. Towards sustainability in world fisheries. Nature 418, 689-695.

PCD, 2004. Plan communale de developpement CR Anororo - FID. Unpublished report, Toamasina, Madagascar. 43 pp.

Persky, J., 1995. Retrospectives: The ethology of *homo economicus*. Journal of Economic Perspectives 9 (2), 221-231.

Peterson, A. M., and Stead, S. M., 2011. Rule breaking and livelihood options in marine protected areas. Environmental Conservation 38 (3), 342-352.

Pidgeon, M., 1996. An ecological survey of Lake Alaotra and selected wetlands of central and eastern Madagascar in analysing the demise of the Madagascar pochard *Aythya innotata*. World Wide Fund for Nature, Antananarivo. 146 pp.

Pierce, S. M., Cowling, R. M., Knight, A. T., Lombard, A. T., Rouget, M., and Wolf, T., 2005. Systematic conservation planning products for land-use planning: Interpretation for implementation. Biological Conservation 125 (4), 441-458.

Pitcher, T. J., 1994. Impact of species changes in African lakes. Final Technical Report (Part 1) Renewable Resources Assessment Group, London, UK. 65 pp.

Pitcher, T. J., Hart, P. J. B., and Pauly, D., editors. 1998. Reinventing fisheries management. Kluwer, London. pp.

Pitt, H., 2007. To live with the sea: Community-based management of marine resources in southwest Madagascar. Independent Study Project, School for International Training. Northwestern University, ISP Collection. Paper 120. 36 pp.

Polasky, S., 2008. Why conservation planning needs socioeconomic data. Proceedings of the National Academy of Sciences 105 (18), 6505-6506.

Polasky, S., Camm, J. D., Solow, A. R., Csuti, B., White, D., and Ding, R., 2000. Choosing reserve networks with incomplete species information. Biological Conservation 94 (1), 1-10.

Pollnac, R., Christie, P., Cinner, J. E., Dalton, T., Daw, T. M., Forrester, G. E., Graham, N. A. J., and McClanahan, T. R., 2010. Marine reserves as linked social-ecological systems. Proceedings of the National Academy of Sciences 107 (43), 18262-18265.

Pomeroy, R. S., 1995. Community-based and co-management institutions for sustainable coastal fisheries management in Southeast Asia. Ocean & Coastal Management 27 (3), 143-162.

Pomeroy, R. S., and Rivera-Guieb, R., 2005. Fishery co-management: A practical handbook. CABI Publishing and International Development Research Centre, Cambridge. 253 pp.

Pound, B., Snapp, S., McDougall, C., and Braun, A., editors. 2003. Managing natural resources for sustainable livelihoods: Uniting science and participation. Earthscan Publications Ltd, London. 252 pp.

Powers, J. E., and Abeare, S. M., 2009. Fishing effort redistribution in response to area closures. Fisheries Research 99 (3), 216-225.

Powers, J. E., and Monk, M. H., 2010. Current and future use of indicators for ecosystem based fisheries management. Marine Policy 34 (3), 723-727.

Pressey, R. L., and Bottrill, M. C., 2009. Approaches to landscape- and seascape-scale conservation planning: Convergence, contrasts and challenges. Oryx 43 (04), 464-475.

Pressey, R. L., and Tully, S. L., 1994. The cost of *ad hoc* reservation: A case study in western New South Wales. Australian Journal of Ecology 19 (4), 375-384.

Pressey, R. L., Watts, M., Ridges, M., and Barrett, T., 2005. C-Plan conservation planning software: User manual, NSW Department of Environment and Conservation, Prince, J., and Hilborn, R., 1998. Concentration profiles and invertebrate fisheries management. pp. 187-196. *In*: G. S. Jamieson and A. Campbell (eds.), Proceedings of the North Pacific Symposium on Invertebrate Stock Assessment and Management. Canadian Special Publication of Fisheries and Aquatic Sciences, Vol. 125.

R Development Core Team, 2011. R: A language and environment for statistical computing, R version 2.13.1. R Foundation for Statistical Computing, Vienna, Austria.

R Development Core Team, 2012. R: A language and environment for statistical computing, R version 2.14.2. R Foundation for Statistical Computing, Vienna, Austria.

Rademeyer, R. A., Plaganyi, E. E., and Butterworth, D. S., 2007. Tips and tricks in designing management procedures. ICES Journal of Marine Science 64, 618-625.

Ralainasolo, F. B., 2004. Effets des actions anthropiques sur la dynamique de la population de *Hapalemur griseus alaotrensis* (Rumpler, 1975) dans son habitat naturel. Diplome d'Etude Approfondie (DEA). Universite d'Antananarivo, Antananarivo. 84 pp.

Ralainasolo, F. B., Waeber, P. O., Ratsimbazafy, J., Durbin, J., and Lewis, R., 2006. The Alaotra gentle lemur: Population estimation and subsequent implications. Madagascar Conservation and Development 1 (1), 9-10.

Ramanampamonjy, J. R., Rasoamampionona Raminosoa, N., Andrianandrasana, H. T., Wilmé, L., Durbin, J., Lewis, R. E., Razafindrajao, F., Razafindramahatra, L., Randriamanampisoa, H., Randriamahefasoa, J., and Rakotoniaina, L. J., 2003. Information Sheet on Ramsar Wetlands: Lake Alaotra Wetlands and Catchment Basin. Durrell Wildlife Conservation Trust Madagascar, Antananarivo. 12 pp.

Ranaivonasy, J., Raharinjanahary, H., and Durbin, J., 2005. Etude de l'évolution des différents régimes de gestion des aires protégées à Madagascar (A Review of Development of Different Management Regimes for the New Protected Areas in Madagascar). Etude de cas : la future aire protégée de l'Alaotra. Unpublished report. Durrell Wildlife Conservation Trust, Antananarivo, Madagascar. 12 pp.

Randall, S., and Koppenhaver, T., 2004. Qualitative data in demography: The sound of silence and other problems. Demographic Research 11 (3), 57-94.

Ratner, B. D., Ha, D. T., Kosal, M., Nissapa, A., and Chanphengxay, S., 2004. Undervalued and overlooked: Sustaining rural livelihoods through better governance of wetlands. WorldFish Center Studies and Reviews No. 28, WorldFish Center. 24 pp.

Razanadrakoto, D. R., and Rafaliarison, J., 2005. Délimitation des zones de frai dans le lac Alaotra. Durrell Wildlife Conservation Trust, Antananarivo. 45 pp.

Rebelo, L. M., McCartney, M., and Finlayson, C., 2010. Wetlands of Sub-Saharan Africa: Distribution and contribution of agriculture to livelihoods. Wetlands Ecology and Management 18 (5), 557-572.

Redman, C. L., 1999. Human dimensions of ecosystem studies. Ecosystems 2 (4), 296-298.

Redpath, S. M., Arroyo, B. E., Leckie, F. M., Bacon, P., Bayfield, N., Gutierrez, R. J., and Thirgood, S. J., 2004. Using decision modeling with stakeholders to reduce human-wildlife conflict: A raptor-grouse case study. Conservation Biology 18 (2), 350-359.

Reed, M. S., 2008. Stakeholder participation for environmental management: A literature review. Biological Conservation 141 (10), 2417-2431.

Reinthal, P. N., and Stiassny, M. L. J., 1991. The freshwater fishes of Madagascar: A study of an endangered fauna with recommendations for a conservation strategy. Conservation Biology 5 (2), 231-243.

Reyers, B., Rouget, M., Jonas, Z., Cowling, R. M., Driver, A., Maze, K., and Desmet, P., 2007. Developing products for conservation decision-making: Lessons from a spatial biodiversity assessment for South Africa. Diversity and Distributions 13 (5), 608-619.

Reyers, B., Roux, D. J., Cowling, R. M., Ginsburg, A. E., Nel, J. L., and O'Farrell, P., 2010. Conservation planning as a transdisciplinary process. Conservation Biology 24 (4), 957-965.

Richardson, E. A., Kaiser, M. J., Edwards-Jones, G., and Possingham, H. P., 2006. Sensitivity of marine-reserve design to the spatial resolution of socioeconomic data. Conservation Biology 20 (4), 1191-1202.

Rist, J., Rowcliffe, M., Cowlishaw, G., and Milner-Gulland, E. J., 2008. Evaluating measures of hunting effort in a bushmeat system. Biological Conservation 141 (8), 2086-2099.

Rist, L., Shaanker, R. U., Milner-Gulland, E. J., and Ghazoul, J., 2010. The use of traditional ecological knowledge in forest management: An example from India. Ecology and Society 15 (1), 3.

Roberts, C. M., Bohnsack, J. A., Gell, F., Hawkins, J. P., and Goodridge, R., 2001. Effects of marine reserves on adjacent fisheries. Science 294 (5548), 1920-1923.

Robinson, J. G., and Bennett, E. L., 2004. Having your wildlife and eating it too: An analysis of hunting sustainability across tropical ecosystems. Animal Conservation 7 (4), 397-408.

Rupp, D. E., Wainwright, T. C., Lawson, P. W., and Peterson, W. T., 2012. Marine environment-based forecasting of coho salmon (*Oncorhynchus kisutch*) adult recruitment. Fisheries Ocenaography 21 (1), 1-19.

Russ, G. R., Stockwell, B., and Alcala, A. C., 2005. Inferring versus measuring rates of recovery in no-take marine reserves. Marine Ecology Progress Series 292, 1-12.

Ruttan, L., 1998. Closing the commons: Cooperation for gain or restraint? Human Ecology 26 (1), 43-66.

Sainsbury, K. J., Punt, A. E., and Smith, A. D. M., 2000. Design of operational management strategies for achieving fishery ecosystem objectives. ICES Journal of Marine Science 57 (3), 731-741.

Salas, S., Chuenpagdee, R., Seijo, J. C., and Charles, A., 2007. Challenges in the assessment and management of small-scale fisheries in Latin America and the Caribbean. Fisheries Research 87 (1), 5-16.

Salas, S., and Gaertner, D., 2004. The behavioural dynamics of fishers: Management implications. Fish and Fisheries 5, 153-167.

Sale, P. F., Cowen, R. K., Danilowicz, B. S., Jones, G. P., Kritzer, J. P., Lindeman, K. C., Planes, S., Polunin, N. V. C., Russ, G. R., Sadovy, Y. J., and Steneck, R. S., 2005. Critical science gaps impede use of no-take fishery reserves. Trends in Ecology & Evolution 20 (2), 74-80.

Sarkar, S., Pressey, R. L., Faith, D. P., Margules, C. R., Fuller, T., Stoms, D. M., Moffett, A., Wilson, K. A., Williams, K. J., Williams, P. H., and Andelman, S., 2006. Biodiversity conservation planning tools: Present status and challenges for the future. Annual Review of Environment and Resources 31, 123-159.

Schwartzman, S., Moreira, A., and Nepstad, D., 2000. Rethinking tropical forest conservation: Perils in parks. Conservation Biology 14 (5), 1351-1357.

SEAFDEC, 2005. Handbook on collecting fishery statistics for inland and coastal fisheries. Southeast Asian Fisheries Development Center, Bangkok. 165 pp.

Seijo, J. C., Defeo, O., and Salas, S., 1998. Fisheries bioeconomics: Theory, modelling and management. FAO Fisheries Technical Paper No. 368, FAO, Rome. 108 pp.

Seixas, C. S., and Begossi, A., 1998. Do fishers have territories? Use of fishing grounds at Aventureiro (Ilha Grande, Brazil). *in* Crossing Boundaries, Seventh Conference of the International Association for the Study of Common Property, Burnaby, Canada.

Shertzer, K. W., and Prager, M. H., 2007. Delay in fishery management: Diminished yield, longer rebuilding, and increased probability of stock collapse. ICES Journal of Marine Science 64 (1), 149-159.

Simon, H. A., 1955. A behavioral model of rational choice. The Quarterly Journal of Economics 69 (1), 99-118.

Singer, E., Von Thurn, D. R., and Miller, E. R., 1995. Confidentiality assurances and response: A quantitative review of the experimental literature. Public Opinion Quarterly 59 (1), 66-77.

Sitati, N. W., and Walpole, M. J., 2006. Assessing farm-based measures for mitigating human-elephant conflict in Transmara District, Kenya. Oryx 40 (3), 279-286.

Smith, A. D. M., Sainsbury, K. J., and Stevens, R. A., 1999. Implementing effective fisheries-management systems - management strategy evaluation and the Australian partnership approach. ICES Journal of Marine Science 56 (6), 967-979.

Smith, D. A., 2003. Participatory mapping of community lands and hunting yields among the Bugle of Western Panama. Human Organization 62 (4), 332-343.

Smith, L. E. D., Nguyen Khoa, S., and Lorenzen, K., 2005. Livelihood functions of inland fisheries: Policy implications in developing countries. Water Policy 7 (4), 359-383.

Smith, M. D., 2005. State dependence and heterogeneity in fishing location choice. Journal of Environmental Economics and Management 50 (2), 319-340.

Smith, M. D., and Wilen, J. E., 2003. Economic impacts of marine reserves: The importance of spatial behavior. Journal of Environmental Economics and Management 46 (2), 183-206.

Smith, M. D., and Zhang, J., 2007. Sorting models in discrete choice fisheries analysis. American Agricultural Economics Association Annual Meeting, 29 July - 1 August, 2007. Portland, Oregon. p. 31.

Smith, R. J., 2004. Conservation Land-Use Zoning (CLUZ) software, Durrell Institute of Conservation and Ecology, Canterbury, UK.

Smith, R. J., Goodman, P. S., and Matthews, W. S., 2006. Systematic conservation planning: A review of perceived limitations and an illustration of the benefits, using a case study from Maputaland, South Africa. Oryx 40 (04), 400-410.

Smith, R. J., Minin, E. D., Linke, S., Segan, D. B., and Possingham, H. P., 2010. An approach for ensuring minimum protected area size in systematic conservation planning. Biological Conservation 143 (11), 2525-2531.

Sodhi, N. S., Brook, B. W., and Bradshaw, C. J. A., 2007. Tropical Conservation Biology. Blackwell Publishing Ltd, Oxford. 332 pp.

Sommerville, M., Jones, J. P. G., Rahajaharison, M., and Milner-Gulland, E. J., 2010a. The role of fairness and benefit distribution in community-based Payment for Environmental Services interventions: A case study from Menabe, Madagascar. Ecological Economics 69 (6), 1262-1271.

Sommerville, M., Milner-Gulland, E. J., Rahajaharison, M., and Jones, J. P. G., 2010b. Impact of a community-based payment for environmental services intervention on forest use in Menabe, Madagascar. Conservation Biology 24 (6), 1488-1498.

Songorwa, A. N., 1999. Community-based wildlife management (CWM) in Tanzania: Are the communities interested? World Development 27 (12), 2061-2079.

Soulé, M. E., and Sanjayan, M. A., 1998. ECOLOGY: Conservation Targets: Do They Help? Science 279 (5359), 2060-2061.

St. John, F. A. V., Edwards-Jones, G., Gibbons, J. M., and Jones, J. P. G., 2010. Testing novel methods for assessing rule breaking in conservation. Biological Conservation 143 (4), 1025-1030.

Stewart, R. R., Noyce, T., and Possingham, H. P., 2003. Opportunity cost of ad hoc marine reserve design decisions: An example from South Australia. Marine Ecology Progress Series 253, 25-38.

Stewart, R. R., and Possingham, H. P., 2005. Efficiency, costs and trade-offs in marine reserve system design. Environmental Modeling and Assessment 10 (3), 203-213.

Stigler, G. J., and Becker, G. S., 1977. De gustibus non est disputandum. The American Economic Review 67 (2), 76-90.

Suski, C. D., and Cooke, S. J., 2007. Conservation of aquatic resources through the use of freshwater protected areas: Opportunities and challenges. Biodiversity and Conservation 16, 2015-2029.

Sutherland, W. J., 2000. The conservation handbook: Research, management and policy. Blackwell Science, Oxford. 278 pp.

Sutinen, J. G., 1999. What works well and why: Evidence from fishery-management experiences in OECD countries. ICES Journal of Marine Science 56, 1051-1058.

Sverdrup-Jensen, S., and Raakjær Nielsen, J., 1998. Co-management in small-scale fisheries: A synthesis of Southern and West African experiences. Institute for Fisheries Management and Coastal Community Development (IFM), The North Sea Centre, Denmark. 23 pp.

Swain, D. P., and Wade, E. J., 2003. Spatial distribution of catch and effort in a fishery for snow crab (*Chionoecetes opilio*): Tests of predictions of the ideal free distribution. Canadian Journal of Fisheries and Aquatic Sciences 60 (8), 897-909.

Teh, L., Zeller, D., Cabanban, A., and Sumaila, U., 2007. Seasonality and historic trends in the reef fisheries of Pulau Banggi, Sabah, Malaysia. Coral Reefs 26 (2), 251-263.

Thaler, R. H., 1988. Anomalies: The ultimatum game. Journal of Economic Perspectives 2 (4), 195-206.

Thaler, R. H., 2000. From Homo economicus to Homo sapiens. Journal of Economic Perspectives 14 (1), 133-141.

Ticheler, H. J., Kolding, J., and Chanda, B., 1998. Participation of local fishermen in scientific fisheries data collection: A case study from the Bangweulu Swamps, Zambia. Fisheries Management and Ecology 5 (1), 81-92.

Tidd, A. N., Hutton, T., Kell, L. T., and Padda, G., 2011. Exit and entry of fishing vessels: An evaluation of factors affecting investment decisions in the North Sea English beam trawl fleet. ICES Journal of Marine Science 68 (5), 961-971.

Tupper, M. H., Wickstrom, K., Hilborn, R., Roberts, C. M., Bohnsack, J. A., Gell, F., Hawkins, J. P., and Goodridge, R., 2002. Marine reserves and fisheries management. Science 295 (5558), 1233-1235.

Tversky, A., and Kahneman, D., 1974. Judgment under uncertainty: Heuristics and biases. Science 185 (4157), 1124-1131.

Uychiaoco, A. J., Arceo, H. O., Green, S. J., De La Cruz, M. T., Gaite, P. A., and Aliño, P. M., 2005. Monitoring and evaluation of reef protected areas by local fishers in the Philippines: Tightening the adaptive management cycle. Biodiversity and Conservation 14 (11), 2775-2794.

Valcic, B., 2009. Spatial policy and the behaviour of fishermen. Marine Policy 33 (2), 215-222.

van Oostenbrugge, J. A. E., van Densen, W. L. T., and Machiels, M. A. M., 2001. Risk aversion in allocating fishing effort in a highly uncertain coastal fishery for pelagic fish, Moluccas, Indonesia. Canadian Journal of Fisheries and Aquatic Sciences 58 (8), 1683-1691.

van Poorten, B. T., Arlinghaus, R., Daedlow, K., and Haertel-Borer, S. S., 2011. Social-ecological interactions, management panaceas, and the future of wild fish populations. Proceedings of the National Academy of Sciences 108 (30), 12554-12559.

Vanden Bossche, J.-P., and Bernacsek, G. M., 1991. Source book for the inland fishery resources of Africa: Vol. 1. CIFA Technical Paper. No. 18.1, FAO, Rome. 240 pp.

Venables, W. N., and Dichmont, C. M., 2004a. A generalised linear model for catch allocation: An example from Australia's Northern Prawn Fishery. Fisheries Research 70 (2-3), 409-426.

Venables, W. N., and Dichmont, C. M., 2004b. GLMs, GAMs and GLMMs: An overview of theory for applications in fisheries research. Fisheries Research 70 (2-3), 319-337.

Vriend, N. J., 1996. Rational behavior and economic theory. Journal of Economic Behavior and Organization 29 (2), 263-285.

Waeber, P. O., and Hemelrijk, C. K., 2003. Female dominance and social structure in Alaotran gentle lemurs. Behaviour 140 (10), 1235-1246.

Walters, B. B., 2004. Local management of mangrove forests in the Philippines: Successful conservation or efficient resource exploitation? Human Ecology 32 (2), 177-195.

Walters, C., 1997. Challenges in adaptive management of riparian and coastal ecosystems. Conservation Ecology 1 (2), 1 [online] URL: http://www.consecol.org/vol1/iss2/art1/.

Watts, M. E., Ball, I. R., Stewart, R. S., Klein, C. J., Wilson, K., Steinback, C., Lourival, R., Kircher, L., and Possingham, H. P., 2009. Marxan with Zones: Software for optimal conservation based land- and sea-use zoning. Environmental Modelling & Software 24 (12), 1513-1521.

Weeks, R., Russ, G. R., Alcala, A. C., and White, A. T., 2010a. Effectiveness of marine protected areas in the Philippines for biodiversity conservation. Conservation Biology 24 (2), 531-540.

Weeks, R., Russ, G. R., Bucol, A. A., and Alcala, A. C., 2010b. Incorporating local tenure in the systematic design of marine protected area networks. Conservation Letters 3 (6), 445-453.

Weeks, R., Russ, G. R., Bucol, A. A., and Alcala, A. C., 2010c. Shortcuts for marine conservation planning: The effectiveness of socioeconomic data surrogates. Biological Conservation 143 (5), 1236-1244.

Weladji, R. B., Moe, S. R., and Veveld, P., 2003. Stakeholder attitudes towards wildlife policy and the Benoue Wildlife Conservation Area, North Cameroon. Environmental Conservation 30 (4), 334-343.

Welcomme, R. L., 1985. River fisheries. FAO Fisheries Technical Paper. No. 262, FAO, Rome. 330 pp.

Welcomme, R. L., 1999. A review of a model for qualitative evaluation of exploitation levels in multi-species fisheries. Fisheries Management and Ecology 6 (1), 1-19.

Welcomme, R. L., 2001. Inland Fisheries: Ecology and Management. Fishing News Books, Blackwell Science, UK. 358 pp.

Welcomme, R. L., 2003. Data requirements for inland fisheries management. pp. 59-67. *In*: T. Clayton, S. Funge-Smith, D. Bartley, and C. Barlow (eds.), New approaches for the improvement of inland capture fishery statistics in the Mekong Basin. Food and Agricultural Organization of the United Nations and Mekong River Commission, Erewan Press, Bangkok, Thailand, Udon Thani, Thailand.

Welcomme, R. L., 2011a. An overview of global catch statistics for inland fish. ICES Journal of Marine Science 68 (8), 1751-1756.

Welcomme, R. L., 2011b. Review of the state of world fishery resources: Inland fisheries. FAO Fisheries and Aquaculture Circular. No. 942, Rev. 2, FAO Inland Water Resources and Aquaculture Service, Fishery Resources Division, Rome. 110 pp.

Welcomme, R. L., Cowx, I. G., Coates, D., Béné, C., Funge-Smith, S., Halls, A., and Lorenzen, K., 2010. Inland capture fisheries. Philosophical Transactions of the Royal Society B: Biological Sciences 365, 2881-2896.

Wells, M., Brandon, K., and Hannah, L., 1992. People and parks: Linking protected area management with local communities. The World Bank, Washington, DC, USA. 116 pp.

Western, D., and Wright, M., editors. 1994. Natural connections: Perspectives in community-based conservation. Island Press, Washington, DC, USA. pp.

Wiber, M., Berkes, F., Charles, A., and Kearney, J., 2004. Participatory research supporting community-based fishery management. Marine Policy 28 (6), 459-468.

Wilen, J. E., Smith, M. D., Lockwood, D., and Botsford, L. W., 2002. Avoiding surprises: Incorporating fisherman behavior into management models. Bulletin of Marine Science 70 (2), 553-575.

Wilmé, A. L., 1994. Status, distribution and conservation of two Madagascar bird species endemic to Lake Alaotra: Delacour's Grebe *Tachybaptus rufolavatus* and Madagascar pochard *Aythya innotata*. Biological Conservation 69 (1), 15-21.

Wilson, J., 2006. Matching social and ecological systems in complex ocean fisheries. Ecology and Society 11, 9.

Wilson, K. A., Westphal, M. I., Possingham, H. P., and Elith, J., 2005. Sensitivity of conservation planning to different approaches to using predicted species distribution data. Biological Conservation 122 (1), 99-112.

Winkler, R., 2011. Why do ICDPs fail?: The relationship between agriculture, hunting and ecotourism in wildlife conservation. Resource and Energy Economics 33 (1), 55-78.

Wise, L., Murta, A. G., Carvalho, J. P., and Mesquita, M., 2012. Qualitative modelling of fishermen's behaviour in a pelagic fishery. Ecological Modelling 228 (0), 112-122.

Wood, S. N., editor. 2006. Generalized additive models: An introduction with R. Chapman & Hall / CRC, Taylor & Francis Group, London. 392 pp.

Wood, S. N., 2008. Fast stable direct fitting and smoothness selection for generalized additive models. Journal of the Royal Statistical Society: Series B (Statistical Methodology) 70 (3), 495-518.

Woodroffe, R., Thirgood, S., and Rabinowitz, A., editors. 2005. People and wildlife: Conflict or coexistence? Cambridge University Press, Cambridge. 497 pp.

World Bank, 2008. World development indicators: Poverty data. The World Bank, Washington, D.C. 32 pp.

World Commission on Environment and Development, 1987. Our common future. United Nations, Oxford, Oxford University Press. 27 pp.

Yoccoz, N. G., Nichols, J. D., and Boulinier, T., 2001. Monitoring of biological diversity in space and time. Trends in Ecology & Evolution 16 (8), 446-453.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., and Smith, G. M., 2009. Mixed effects models and extensions in ecology with R. Springer, New York. 574 pp.

APPENDIX S3 – SUPPORTING INFORMATION FOR CHAPTER 3

Appendix S3.1. Catch-monitoring interview datasheet

Catch-monitoring interview – page 1

CATCH INTERVIEW

Date: ____________ Time: AM / PM

Interview site: 1 2 Interviewer: ____________

Fisher: ____________ Number of fishers: ____________

Landing site: ____________ Followed today: Yes / No

Fishing locations: ____________

____

Travel time and distance to fishing location: Time ______ KM ______

Travel time and distance from fishing location: Time ______ KM ______

Pirogue used: Yes / No Number of pirogues: ______

Fishing method used:

TRAP ☐ Number used: ______

Time spent fishing: ______

Time interval (placed & retrieved): ______

Bait used: Yes / No

Time to collect bait: ______

Cost of bait: ______ Ariary

Length of trap: ______

Diameter of trap: ______

Mesh size: ______ mm

Mesh material: Line Bamboo Nylon

GILL NET ☐ Number used: ______

Time spent fishing: ______

Number of times fish retrieved: ______

Time interval (placed & retrieved): ______

Length of net: ______

Height of net: ______

Mesh size: ______ mm

CAST NET ☐

Time spent fishing: ______

Number of net throws: ______

Radius of net: ______

Mesh size: ______ mm

Bait used: Yes / No

Cost of bait: ______ Ariary

OTHER ☐ ______

Description: ______

Number used: ______

Size: ______ Mesh: ______ mm

Time spent fishing: ______

Number of times fish retrieved: ______

Time interval: ______

Fish captured: Yes / No Sell fish: Yes / No

Where sold: Sandenda Vazaha kely Bicycle collector Local market Not decided

Fish kept: Yes / No Why kept: ______

Number & species kept: ______

Catch-monitoring interview – overleaf

Fish Measured:

ALL or EVERY 2 3 4 5 6 7 8 9 10 ______

	Species	Length (mm)	Kept (✓)		Species	Length (mm)	Kept (✓)
1				26			
2				27			
3				28			
4				29			
5				30			
6				31			
7				32			
8				33			
9				34			
10				35			
11				36			
12				37			
13				38			
14				39			
15				40			
16				41			
17				42			
18				43			
19				44			
20				45			
21				46			
22				47			
23				48			
24				49			
25				50			

Number of fish remaining after measuring: ______

Notes: ______

Appendix S3.2. Semi-structured background interview

SEMI-STRUCTURED INTERVIEW

Introduction (specific phrasing by assistant to cover the following points):
- I am a university student gathering information about fishing activity around Anororo to understand how fishers fish and to identify problems and costs for fishers.
- I am asking questions about your fishing experience, the livelihood that you make from fishing, and the costs involved in doing this. All information will be kept confidential.
- The interview is expected to take approximately 1 hour.
- Please do not answer any question that you do not wish to answer.
- If you have any questions, please ask me at any time.

Background information & demographics:

1. DATE ____________ TIME __________ ASSISTANT ___________
2. NAME OF FISHER ______________________________
3. GENDER Male / Female
4. ETHNIC GROUP Sihanaka / Other ______________
5. AGE CATEGORY < 15 / 15-24 / 25-34 / 35-44 / 45-54 / 55+
6. LEVEL OF EDUCATION ATTENDED None / Primary / Secondary / College or University
7. YEARS LIVED IN ANORORO __________ years
   - If <5 years, where did you live before? ______________________
   - Why did you move to Anororo? ______________________

   ____
8. NUMBER OF PEOPLE CURRENTLY LIVING IN HOUSEHOLD _____ Adults _____ Children
9. OTHER PEOPLE SUPPORTED OUTSIDE OF HOUSEHOLD ______________________
10. PROPERTY OWNED (INDICATE QUANTITIES; IF QUANTITY UNKNOWN TICK [✓] BOX)

ITEM	NUMBER	ITEM	NUMBER
Concrete house		Bicycle	
Traditional house		Zebu	
Rice field	_____ ha	Chickens	
Shop / store		Ducks	
Kibota		Geese	
Motorcycle		Goats or sheep	
Generator		Other:	
Television			
Radio			

Livelihood activities & income:

11. PRIMARY OCCUPATION (FIVELOMANA) ____________________

12. WHAT ACTIVITIES DO YOU DO TO MAKE A LIVING DURING EACH SEASON?

- Rank each in terms of reliance for livelihood.
- Indicate proportion of work time allocated to each activity (20 marbles, each = 5% time).
- For activities other than fishing, estimate income after expenses.

WET SEASON (December to April)

ACTIVITY	YES [✓]	RANK	% TIME	INCOME
Fishing				Total Ariary per
Rice cultivation				Tonnes or Ariary per
Shop / store				Ariary per
Work for someone else				Ariary per
Collect / make marsh products				Ariary per
Other:				Amount and frequency:

DRY SEASON (May to November)

ACTIVITY	YES [✓]	RANK	% TIME	INCOME
Fishing				Total Ariary per
Rice cultivation				Tonnes or Ariary per
Shop / store				Ariary per
Work for someone else				Ariary per
Collect / make marsh products				Ariary per
Other:				Amount and frequency:

Fishing experience & costs:

13. YEARS OF FISHING EXPERIENCE ____________ years

- If <5 years, what did you do before? ____________________

____

Semi-structured Interview – Page 2

14. FISHING FREQUENCY ______ per day / _____ per week / ______ per month / ________ per year

- Why do you fish at this frequency? ____

____

15. PRIMARY FISHING EQUIPMENT OWNED

	NUMBER	COST (EACH)	SIZE
Trap			Length ______; Diameter ______; Mesh______ mm
Gill net			Length ______; Height ______; Mesh ______ mm
Cast net			Radius ________; Mesh _________ mm
Other method:			
Canoe			Capacity __________ persons

16. OTHER FISHING EQUIPMENT OWNED

ITEM	NUMBER	COST (EACH)
Paddle		
Chair		
Lafika		
Salpo		
Dima		
Lamp		
Other:		

ITEM	NUMBER	COST (EACH)
Pole		
Antsymailoka		
Fibarana		
Antsibe		
Pliers		
Lasitra		
Other:		

17. COST OF BATTERIES FOR FISHING LAMP Ariary ____________ per ____________

18. COST OF FISHING LINE FOR REPAIRS Ariary ____________ per ____________

19. DO YOU PURCHASE A FISHING LICENSE? Yes / No

- If yes, what is the cost and how often? Ariary ____________ per ____________

20. DO YOU RENT OR BORROW ANY FISHING EQUIPMENT THAT YOU USE? Yes / No

- If yes, what equipment and at what cost? ________________________________

____

21. DO YOU USE BAIT WHEN YOU FISH? Yes / No

- If yes, how often? ____
- If bought, what is the cost? ________________________________
- If gathered, how much time to collect? ___________________________

Semi-structured Interview – Page 3

Fishing behaviour & catches:

22. WHICH FISHING LOCATIONS DO YOU USE?

LOCATION	TRAVEL TIME BETWEEN LANDING SITE AND FISHING LOCATION	MONTH(S) FISHED

- Why do you fish at these locations? ______________________

23. FISH IN SAME LOCATION(S) OVER THE LAST 5 YEARS Yes / No

IF YES

- Why have you stayed in the same locations? ______________________

IF NO

- Where did you fish before? ______________________
- Why did you change locations? ______________________

24. QUANTITY OF FISH CAPTURED (FOR EACH SEASON)

WET SEASON	Method Used: Number of Fish	OR Bucketfuls	Proportion of good, average, and bad days (20 marbles)
Good day			
Average day			
Bad day			

DRY SEASON	Method Used: Number of Fish	OR Bucketfuls	Proportion of good, average, and bad days (20 marbles)
Good day			
Average day			
Bad day			

25. OVER THE LAST 5 YEARS CATCHES HAVE Increased / Same / Decreased

- Why? ____________________

26. OVER THE LAST 5 YEARS FISH SIZES HAVE Increased / Same / Decreased

- Why? ____________________

27. WHY DO YOU USE THE METHOD THAT YOU USE? ____________________

28. HAVE YOU FISHED WITH THE SAME METHOD OVER THE LAST 5 YEARS? Yes / No

- If no, what method did you use before? ____________________
- Why did you change methods? ____________________

29. NUMBER OF PEOPLE FISHING NOW COMPARED TO:

- When you began fishing More / Same / Less By how many? ________
- 5 years ago More / Same / Less By how many? ________

30. DO YOU WANT TO CONTINUE BEING A FISHER? Yes / No

- Why? ____________________

31. DO YOU WANT YOUR SON TO BE A FISHER? Yes / No

- Why? ____________________

Views about the marsh and fishing regulations:

32. DO YOU USE THE MARSH? Yes / No

- If yes, why? (Tick [✓] all stated and circle the factor nominated as most used)

Fishing		Medicinal plants	
Source of water		Hunting wildlife (e.g., bandro, birds)	
Construction materials		Enjoyment	
Growing rice		Other:	

33. IS THE MARSH VALUABLE FOR OTHER REASONS? Yes / No

- If yes, why? (Tick [✓] all stated and circle the factor nominated as most valuable)

Fish habitat		Medicinal plants	
Habitat for wildlife		Hunting wildlife (e.g., bandro, birds)	
Construction materials		Enjoyment	
Growing rice		Climate	
Water & air quality		Other:	

Semi-structured Interview – Page 5

34. ARE THERE REGULATIONS CONTROLLING FISHING IN THE LAKE OR MARSH? Yes / No

- If yes, who manages these? ________________________________

____

If yes, continue with questions 35 to 41. If no, go to question 40.

35. ARE THERE PLACES WHERE PEOPLE ARE NOT SUPPOSED TO FISH? Yes / No

- If yes, where are they? ________________________________

____

36. ARE THERE FISHING METHODS THAT PEOPLE ARE NOT SUPPOSED TO USE? Yes / No

- If yes, what are they? ________________________________

____

37. WHAT DO YOU THINK ABOUT THE REGULATIONS? ________________________________

____

____

____

38. ARE THERE WAYS IN WHICH THE REGULATIONS ARE GOOD FOR YOU? Yes / No

- If yes, what are they? ________________________________

____

____

39. ARE THERE WAYS IN WHICH THE REGULATIONS ARE BAD FOR YOU? Yes / No

- If yes, what are they? ________________________________

____

____

40. DO YOU HAVE ANY SUGGESTIONS FOR OTHER WAYS TO MANAGE THE FISHERY? Yes / No

- If yes, what are they? ________________________________

____

____

41. DO YOU HAVE ANY QUESTIONS ABOUT THE PROJECT THAT YOU WOULD LIKE TO ASK ME?

____

____

____

____

Appendix S3.3. Semi-structured scenario interview

SEMI-STRUCTURED SCENARIO INTERVIEW

Introduction:

- I am a university student gathering information about fishing activity around Anororo to understand how fishers fish and to identify problems and costs for fishers.
- During the first interview I asked questions about your fishing experience.
- During this interview, I would like to explore how regulations or hypothetical changes in the fishery might affect you or influence how and where you fish.
- All information will be kept confidential.
- The interview is expected to take approximately 60 to 90 minutes.
- Please do not answer any question that you do not wish to answer.
- If you have any questions, please ask me at any time.

DATE ________ TIME ________ ASSISTANT ________

NAME OF FISHER ________ FISHER ID ________

1. How has fishing been during this past year? What are the reasons for this? ________

1a. Do you think the future for fishing in Lake Alaotra will be GOOD / BAD

Why? ________

1b. Do you think there will be any changes over the next 5 years? YES / NO

If yes, what changes? Why? ________

1c. Do you think there will be any changes over the next 20 years? YES / NO

If yes, what changes? Why? ________

2. What would happen to the fishery if there were no regulations? How would this affect you?

____

3. Information about where fishers fish, what gear they use, how long they fish for, the types of fish they catch, and the size of fish caught can be used to manage a fishery. This information can show whether the fish population is growing or declining, the amount of effort fishers use to make their catch, and provide ideas to keep the fishery sustainable. Would you be willing to keep records of your fishing (i.e., where you fish, what gear you use, how long you fish for, and the species and length of fish that you catch) to give to the authorities, if this information was definitely going to be used to manage the fishery better?

YES / NO If no, why not? ________

1

4. Which fishing locations have you been using over the last year? (*compare with locations stated in first interview – ask fisher to explain any differences*).

Locations: ______________________________

____

Reasons for differences: ______________________________

____

4a. Are the locations where you fish *your* fishing territory? YES / NO

______________________________ *If no, go to question 4c.*

4b. How did these locations become your fishing territory? (*e.g., Location handed down in the family like a rice field? Traditional or frequent use? One of the first people to start fishing there? Constructed a hamatra? Replaced other fishers who left the location? Other?*)

____

____

4c. Can other people fish freely at these locations (*i.e., are they open to anyone who wants to fish there*)? YES / NO If no, why not?

____

5. Approximately how many fishers fish in each location that you use? Specify for each location.

Location	**Number of Fishers**	**Location**	**Number of Fishers**

5a. Does this intensity of fishing affect the fish? YES / NO In what way(s)?

____

5b. Is this intensity of fishing good or bad for you? GOOD / BAD In what way(s)?

____

5c. If only half as many fishers used your locations would the quantity of fish you catch change?
YES / NO If yes, INCREASE / DECREASE *and then continue with:*

i. By how much and over what period of time? ______________________________

____

ii. What would you do in response in the first month? ______________________________

____

iii. What would you do after a year? ______________________________

____

2

5d. If twice as many fishers used your locations would the quantity of fish you catch change?
YES / NO If yes, INCREASE / DECREASE *and then continue with:*

i. By how much and over what period of time? ______________________

ii. What would you do in response in the first month? ______________________

iii. What would you do after a year? ______________________

5e. If four times as many fishers used your locations would the quantity of fish you catch change?
YES / NO If yes, INCREASE / DECREASE *and then continue with:*

i. By how much and over what period of time? ______________________

ii. What would you do in response in the first month? ______________________

iii. What would you do after a year? ______________________

6. *If fisher fishes in the lake:* If there were much less tilapia in the lake would you fish more in the marsh?
YES / NO Why? ______________________

If fisher fishes in the marsh: If there were much less snakehead in the marsh would you fish more in the lake? YES / NO Why? ______________________

7. If there was a program to remove water hyacinth and giant salvinia from the lake, marsh, and channels, would you fish in a different location from where you fish now? YES / NO

- If yes, where would you fish instead and why this location? ______________________
- If no, why not? ______________________

7a. Would removing these plants lead to a change in your income? YES / NO

- If yes, INCREASE / DECREASE By how much? ______________________

7b. Would removing these plants be good or bad in other ways? ______________________

8. Are there currently any no-take zones or protected areas in the lake or marsh? YES / NO
If no, go to question 9.

- If yes, where are they? ______________________
- Why are they there? ______________________

8a. Are there any negative impacts or costs of these areas? YES / NO

- If yes, what are they? ______

______

8b. Are there any benefits from these areas? YES / NO

- If yes, what are they? ______

______

8c. Will there be any benefits or costs from these areas in the future? YES / NO

- If yes, what will they be? If no, why? ______

______

9. Where are the best places in the lake or marsh to have no-take zones or protected areas? ______

______

Why? ______

What size should these areas be? ______

What things should the authorities consider when selecting these areas? ______

______

10. If there were well-enforced no-take zones or protected areas in the lake or marsh, do you think the future for fishing in Lake Alaotra over …

… the next 5 years would be BETTER / SAME / WORSE

… the next 20 years would be BETTER / SAME / WORSE

- Would you then want your son to be a fisher? YES / NO
- Why? ______

11. If your fishing locations became no-take zones or protected areas (where all fishing is prohibited), what would you do? (*Circle answers from A to E below, then ask further questions*).

A. Continue fishing there anyway. *Then go to 11a*.
B. Change fishing location. *Then go to 11b*.
C. Stop fishing altogether. *Then go to 11c*.
D. Seek compensation. *Then go to 11d*.
E. Other. *Then go to 11e*.

11a. (*Where the answer was 'A – Continue fishing there anyway' in question 11*).

i. Why would you not change location? ______

______

ii. Would you fish in different ways? ______

______

iii. What would be required for you to stop fishing there? ______

______

4

11b. (*Where the answer was 'B – Change fishing location' in question 11*).

i. Where would you fish?

ii. Would you have to travel further, not as far, or the same as you do now?

MORE / LESS / SAME How much more or less in time and distance?

Time: ____________ Distance: ____________

iii. Would you have to move your fishing equipment? YES / NO

- If yes, would this be EASY / DIFFICULT How much time would be involved to do this? ____________

iv. Would you change fishing methods because of moving location? YES / NO

- If yes, which method and why that method? ____________

____

v. Would you buy or use other equipment because of moving location? (*such as traps, gill nets, cast nets, a larger or smaller pirogue*) YES / NO

- If yes, what equipment and at what cost? ____________

____

vi. Would you spend more, less, or the same time fishing because of moving location?

MORE / LESS / SAME If more or less, by how much? Why? ____________

____

- If less, what would you do instead? ____________

vii. Would you catch more, less, or the same quantity of fish? MORE / LESS / SAME

- If more or less, how much? ____________

viii. Would you catch different species of fish than you catch now? YES / NO

- Describe differences and why. ____________

____

11c. (*Where the answer was 'C – Stop fishing altogether' in question 11*).

i. What would you do instead? ____________

ii. Why would you do this activity? ____________

____

11d. (*Where the answer was 'D – Seek compensation' in question 11*).

i. What form of compensation (*for example, fishing equipment, money*)? ____________

____

ii. How much or what amount of compensation? ____________

iii. From who? ____________

11e. (*Where the answer was 'E – Other' in question 11*).

i. Describe and why? ____________

____

5

12. If your fishing locations became no-take zones …

i. would this lead to a change in your income? YES / NO

- If yes, INCREASE / DECREASE By how much and why?__________

ii. would this change your daily routine? YES / NO

- If yes, how? __________
- Would this be good or bad for you? GOOD / BAD Why?

iii. would this lead to a change in the time you have available to rest or do other activities?

YES / NO If yes, INCREASE / DECREASE

Which activities and by how much? __________

iv. would this affect your life in other ways? YES / NO

- If yes, how? __________

13. Is there a closed period of no fishing each year? YES / NO *If no, go to question 15.*

13a. When is the closed period? __________

13b. What is the purpose of the closed period? __________

13c. Are people still allowed to fish for food during the closed period? YES / NO

13d. Do you fish during the closed period? YES / NO *If no, go to question 13h.*

13e. Do you fish more, the same, or less than normal during the closed period?

MORE / SAME / LESS Why? __________

If less, what do you do instead? __________

13f. Do you fish in different locations than normal during the closed period? YES / NO

If yes, where and why? __________

13g. Do you sell any fish during the closed period? YES / NO

If yes, who do you sell to?__________

Only if the fisher does not fish during the closed period (i.e., the answer to 13d was 'no'):

13h. What do you do instead? Why? __________

14. Are there any negative impacts or costs from of the closed period? YES / NO

- If yes, what are they? __________

14a. Are there any benefits from the closed period? YES / NO

- If yes, what are they? __________

14b. Will there be any benefits or costs from the closed period in the future? YES / NO

- If yes, what will they be?__________
- If no, why?__________

14c. Does the closed period …

i. lead to a change in your income for the year? YES / NO

- If yes, INCREASE / DECREASE By how much and why?__________

ii. change your daily routine? YES / NO

- If yes, how?__________
- Is this good or bad for you? GOOD / BAD Why? __________

iii. lead to a change in the time you have available to rest or do other activities?
YES / NO If yes, INCREASE / DECREASE

- Which activities and by how much?__________

iv. affect your life in other ways? YES / NO

- If yes, how?__________

15. If there has to be a closed period of no fishing each year, what would be the best time of year for this?

- Why at this time?__________
- What should be the duration of the closed period?__________
- Why for this length of time?__________
- Would this closed period be better or worse for you compared to the closed period in question 13? Why?__________

For questions 16 and 17 refer to responses that fisher provided during initial semi-structured interview.

16. If you had twice as many good fishing days (large catches) during the wet season (December to April), what would you do? (*Circle response from A to E below, then ask further questions and write the answers on the lines below*).

A. Fish more. *Then ask*: How and how much more? Why?
B. Fish less and not do anything else (become lazy (kamo)). *Then ask*: Why?
C. Fish less and do something else. *Then ask*: What else would you do? Why?
D. Continue fishing the same as now. *Then ask*: Why?
E. Other. *Then*: Describe and why?

16a. If you had twice as many good fishing days (large catches) during the dry season (May to November), what would you do? (*Circle response from A to E below, then ask further questions and write the answers on the lines below*).

A. Fish more. *Then ask*: How and how much more? Why?

B. Fish less and not do anything else (become lazy (kamo)). *Then ask*: Why?

C. Fish less and do something else. *Then ask*: What else would you do? Why?

D. Continue fishing the same as now. *Then ask*: Why?

E. Other. *Then*: Describe and why?

17. If you had twice as many bad fishing days (small or no catch) during the wet season (December to April), what would you do? (*Circle response from A to G below, then ask further questions and write the answers on the lines below*).

A. Fish more. *Then ask*: How and how much more? Why?

B. Fish less and do something else. *Then ask*: What else would you do? Why?

C. Stop fishing. *Then ask*: What would you do instead? Why?

D. Continue fishing the same as now. *Then ask*: Why?

E. Change fishing location. *Then ask*: Where would you fish? Why?

F. Change fishing gear / method. *Then ask*: What gear / method would you use? Why?

G. Other. *Then*: Describe and why?

17a. If you had twice as many bad fishing days (small or no catch) during the dry season (May to November), what would you do? (*Circle response from A to G below, then ask further questions and write the answers on the lines below*).

A. Fish more. *Then ask*: How and how much more? Why?

B. Fish less and do something else. *Then ask*: What else would you do? Why?

C. Stop fishing. *Then ask*: What would you do instead? Why?

D. Continue fishing the same as now. *Then ask*: Why?

E. Change fishing location. *Then ask*: Where would you fish? Why?

F. Change fishing gear / method. *Then ask*: What gear / method would you use? Why?

G. Other. *Then*: Describe and why?

Thank you very much !

Appendix S3.4. Fisher follow datasheet

FISHER FOLLOW

Date: ____________ Observer: ______________ Landing site: __________

Fisher: ____________________________ Departure time: ______________

Arrival time at first fishing location: ________________

Fishing method used:

TRAP [ ] GILL NET [ ] CAST NET [ ] OTHER [ ] ______________

- For traps = Use counter; take a waypoint at each tenth trap
- For nets = Take a waypoint at each end of each net
- For cast nets = Use counter; take a waypoint each tenth time the cast net is thrown

GPS #: 1 2 3 4 Start GPS: ____ S17° __________ E48° __________

End GPS: ____ S17° __________ E48° __________

Time fishing activity stopped: __________ Counter number: ________

If nets used, number of times nets checked for fish: __________

Return time to landing site: ____________

Landing site: __________ Fish captured: Yes / No

Notes: ____

____

APPENDIX S4 – SUPPORTING INFORMATION FOR CHAPTER 4

Table S4.1. Mean number of fishers who reported fishing within each type of habitat per month during each season. One-sample chi-square tests confirmed significant differences in habitat use between seasons. Data source: background interviews (n = 405).

Habitat	Wet season	Dry season	χ^2	*df*	*p*
Marsh	252	157	22.1	1	< 0.001
Lake-marsh edge	99	191	29.2	1	< 0.001
Lake	12	95	64.4	1	< 0.001
Village border	15	1	12.3	1	< 0.001

Table S4.2. Fishing methods used at Lake Alaotra.

English name	Local name	Description	Frequency of use in Anororo	Legality
Trap	Vovo	Cylindrical trap with bamboo frame. Mesh is monofilament fishing line, nylon, or thin strips of bamboo. Fishers often build a wall of reeds to attach their traps to.	Primary / most common	Legal provided does not build wall of reeds and mesh size $\geq$40mm
Gill net	Harato	Monofilament net with float line and lead line. Mesh woven to fisher specifications. Can be used passively (overnight or while waiting with net) or actively (paddle or pole use to hit water and scare fish into net).	Primary / most common	Legal provided passive use and mesh size $\geq$40mm
Cast net	Épervier or Harato atsipy	Circular net with small weights attached around its circumference. Net is thrown into open water (usually onto bait placed earlier) to spread over the water and sink. Fish are caught as the net is hauled back in.	Common	Legal provided mesh size $\geq$35mm
Line & hook	Lohamandry	A 2m reed rod with 1m of 0.5mm fishing line attached at the centre of the rod. A hook is attached to the end of the line. The hook is usually baited with worms or frogs collected from the marsh.	Common	Legal
Rod & bubble	Jinjira	Hand method. A 2m to 3m reed rod is pushed through the mud at the bottom of the water and a bubble rises when a fish is encountered. The fisher then reaches down to collect the fish by hand. Can be used from a canoe or while wading through shallow water.	Common in dry season	Illegal
Slap & bubble	Mangodo	Hand method. A bat or rod is used to slap the water and scare fish into the mud. Rising bubbles indicate the location of the fish and the fisher then reaches down to collect the fish by hand.	Common in dry season	Illegal
Dip net	Sitra	Mosquito netting (~3mm mesh size) attached to a bamboo frame. One-person dip nets are racket-shaped and used from a canoe. Two-person dip nets are rectangular with a long pocket in the centre to trap fish, and are pulled through water. Both types of dip net are intended to be used to target eastern mosquitofish and/or Madagascar rainbowfish; in practice they are frequently also used to catch juvenile tilapia *spp*.	Somewhat common	Legal for two species: Eastern mosquitofish (*Gambusia holbrooki*) and Madagascar rainbowfish (*Rheocles alaotrensis*)

English name	Local name	Description	Frequency of use in Anororo	Legality
Barbed spear	Leoka	Pole with four barbs attached to one end. Used to kill large fish caught in gill nets and also to catch fish hiding in mud.	Somewhat rare	Illegal when used to catch fish in mud
Dig	Saka	Low water method. People dig or rummage through mud under marsh plants to collect fish (primarily blotched snakehead, *Channa maculata*) by hand.	Somewhat rare	Illegal
Mud enclosure (large)	Valatany	Low water method. Walls of mud are constructed around a relatively large area of marsh to enclose fish. Traps are placed at openings in the wall and fish swim into the traps when water level lowers.	Rare	Illegal
Mud enclosure (small)	Ridrano	Low water method. Walls of mud are constructed around a relatively small area of marsh to enclose fish. Buckets are used to empty the area of water and fish are then collected by hand.	Rare	Illegal
Plant circle	Tosika	Marsh plants are placed to extend to the lake bottom and arranged to create a circular barrier. The plants are then rolled inwards until the only opening is at the centre of the barrier. Small fish become trapped within the barrier and are collected through the opening.	Very rare	Illegal
Drag net*	Harato balle	Similar to a two-person dip net. However, mesh size is larger, approximately 30mm or the width of two adult fingers.	Very rare	Illegal
Seine net*	Harato tarika or Harato be	Standard seine net pulled through the water by a team of 4 to 5 fishers. Nets are usually several hundred metres in length with a mesh size of approximately 40mm.	Very rare	Legal provided mesh size $\geq$40mm
Seine net (large)*	Ramangaoka	An extended seine net pulled through the water by a team of 8 or more fishers. Nets are 1km or more in length with a very small mesh size, often approximately 10mm.	Very rare	Illegal

* Fishing methods reported to be primarily used by fishers based outside of Anororo.

Table S4.3. Mean cost, lifespan, and frequency of repairs for fishing equipment used by Anororo-based fishers; range is shown in parentheses. Data source: background interviews and focus group sessions.

Equipment item	Cost per item (Ariary)[a]	Lifespan	Frequency of repairs
Trap -Monofilament -Nylon -Bamboo	1,884 (200 to 6,000)	5 years 1.5 years 6 months	Every 3 days
Gill net	36,565 (10,000 to 150,000)	4.5 years	Monthly
Cast net	35,111 (16,000 to 100,000)	3 years	Daily
Line & hook	226 (120 to 300)	1.5 months (reed) 1 year (line)	Every 10 days
Dip net	6,500	5 years	n/a
Canoe - wood			Monthly (Repairs for all types of canoes begin after 1 year and occur once a month thereafter)
-Amboramangitra (*Tambourissa*)	100,000	7.5 years	
-Sevalahy (*Buddleia fusca*)	50,000	4 years	
-Mongy (*Croton*)	20,000	1 year	
-Alampona (*Hibiscus*)	35,000	1.5 years	
-Kininina (*Eucalyptus* sp)	35,000	7.5 years	
-Tafonana	40,000	4.5 years	
-Voara	30,000	1 year	
-Arina	40,000	4.5 years	
-Tavôlo	20,000	9 months	
-Vanana	40,000	1.5 years	
-Tapika	100,000	10 years	
-Hazomanitra	40,000	4 years	
-Vatoana	90,000	12 years	
-Rotra	30,000	3 years	
-Amontana	20,000	2 years	
-Volomborona	20,000	2 years	
Paddle -Sevalahy wood	1,856 (300 to 4,000)	2 years	n/a
Pliers -Wood -Aluminum	1,234 (150 to 3,000)	1.75 years 7.5 years	n/a n/a
Headlamp	4,146 (500 to 8,000)	5 months	n/a
Knife	3,155 (350 to 11,000)	2.5 years	n/a
Chair for canoe -Kininina wood	672 (200 to 3,000)	4 years	Annually
Papyrus mat	370 (100 to 3,000)	3 weeks	n/a
Scoop	585 (100 to 1,500)	2 years	n/a

[a] Malagasy Ariary (Ar). At the time of the study GBP£1 = 3,000Ar and USD$1 = 2,080Ar.

Figure S4.1. Fish species observed in Anororo fishers' catches or opportunistically during the study. Species, from left to right, top to bottom are: 1) Nile tilapia (*Oreochromis niloticus niloticus*), 2) Redbreast tilapia (*Tilapia rendalli*), 3) Mozambique tilapia (*Oreochromis mossambicus*), 4) Hybrid tilapia (*O. niloticus niloticus* and *O. macrochir*), 5) Hybrid tilapia (unidentified hybrid; known locally as Lavavava), 6) Blotched snakehead (*Channa maculata*), 7) Goldfish (*Carassius auratus auratus*), 8) Common carp (*Cyprinus carpio carpio*), 9) Eastern mosquitofish (*Gambusia holbrooki*), 10) Madagascar rainbowfish (*Rheocles alaotrensis*), 11) Indonesian short-finned eel (*Anguilla bicolor bicolor*), and 12) Black bass (*Micropterus salmoides*).

APPENDIX S5 – SUPPORTING INFORMATION FOR CHAPTER 5

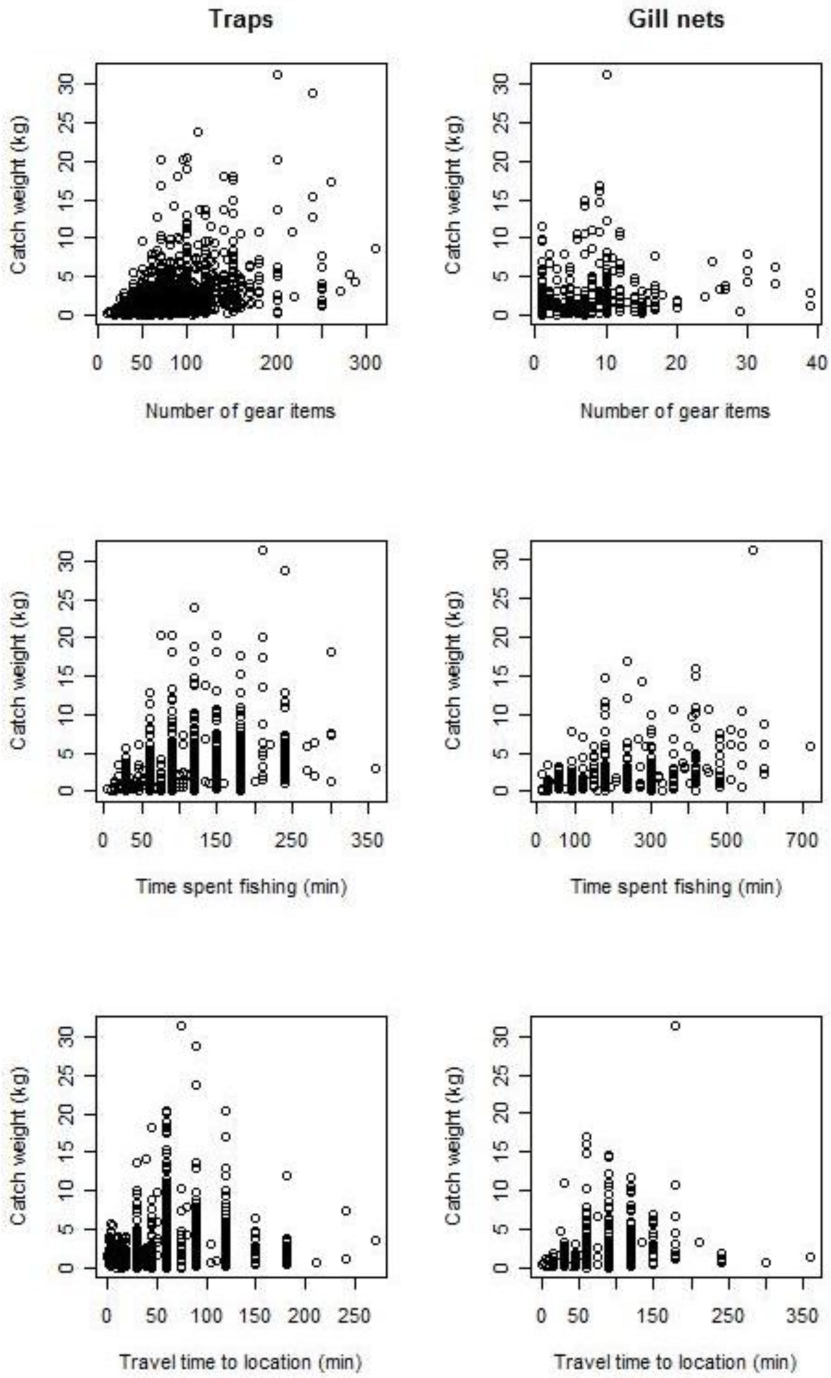


Figure S5.1. Raw catch weight (kg) for each measure of fisher effort by gear type. Catch weight generally increases with increasing fisher effort measured as number of gear items used, time spent fishing, and time spent travelling to a fishing location.

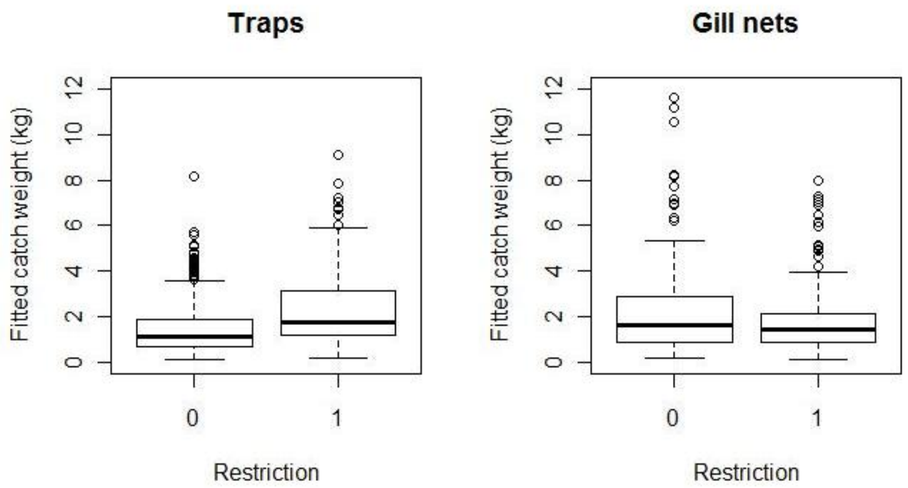


Figure S5.2. Fitted data for the top model based on lowest AIC for each gear type. Box and whisker plots of fitted catch weight (kg) by protected area status (0 = non-restricted; 1 = restricted); the horizontal bar represents the 50th percentile, the top of the box the 75th percentile, and the base of the box the 25th percentile. Whiskers represent the range of data, and open circles are outliers (one outlier was outside the scale).

Table S5.1. Averaged model parameters explaining catch size for trap and gill net fishers. The coefficients, standard error, and lower and upper confidence intervals for each variable are provided for each averaged set of models. Baseline levels for restricted, time period, and habitat variables for both models are 'non-restricted', 'TimePeriod2009_May-Jun', and 'edge', respectively.

Model	Coefficient	SE	Lower CI	Upper CI
Traps				
(Intercept)	3.2100	0.6380	1.960	4.460
Restricted1	0.2610	0.0812	0.102	0.420
TimePeriod2009_Jul-Sep	-0.4050	0.1850	-0.769	-0.042
TimePeriod2009_Oct-Nov	0.2460	0.1800	-0.106	0.598
TimePeriod2010_Jan-Feb	0.2010	0.1670	-0.126	0.528
TimePeriod2010_Mar-Apr	-0.2110	0.1610	-0.527	0.106
TimePeriod2010_May-Jun	-0.4310	0.1630	-0.751	-0.111
TimePeriod2010_Jul-Sep	-0.3650	0.1590	-0.677	-0.053
TimePeriod2010_Oct-Nov	-0.1930	0.1740	-0.533	0.148
log(EstNumberUsed)	0.7050	0.1030	0.503	0.906
log(EstTimeFishing)	0.2680	0.0694	0.132	0.404
log(EstTimeTo)	0.0374	0.0484	-0.057	0.132
log(Size)	0.0694	0.0757	-0.079	0.218
HabitatM	-0.0418	0.0779	-0.195	0.111
log(Mesh)	-0.0263	0.1250	-0.272	0.219
Gill nets				
(Intercept)	2.9800	1.0700	0.874	5.080
Restricted1	-0.0200	0.0762	-0.169	0.129
TimePeriod2009_Jul-Sep	0.3810	0.4450	-0.492	1.250
TimePeriod2009_Oct-Nov	1.0200	0.4420	0.153	1.880
TimePeriod2010_Jan-Feb	-0.8210	0.4680	-1.740	0.096
TimePeriod2010_Mar-Apr	-0.2500	0.4470	-1.130	0.627
TimePeriod2010_May-Jun	-0.0665	0.4480	-0.944	0.811
TimePeriod2010_Jul-Sep	-0.1420	0.4310	-0.987	0.703
TimePeriod2010_Oct-Nov	-0.2500	0.4480	-1.130	0.627
log(EstNumberUsed)	0.2550	0.0761	0.105	0.404
log(EstTimeFishing)	0.2570	0.0849	0.091	0.424
log(EstTimeTo)	0.3670	0.1030	0.165	0.568
log(Size)	0.1720	0.1530	-0.128	0.471
HabitatL	0.0280	0.0920	-0.152	0.208
HabitatM	-0.0727	0.1650	-0.397	0.252
log(Mesh)	0.0650	0.2440	-0.413	0.543

Table S5.2. Mean catch size per trip for all catches and under spatial and temporal restrictions for each gear type. 'Current overall' shows the mean catch averaged over all locations and times of year. 'Spatial closure' compares catches in planned restricted areas with catches in non-restricted fishing locations. 'Temporal closure' compares catches during an October-November closed period with catches over the remainder of the year. Mean catches in non-restricted areas and not in the closed period comprise 'both closures'. Catch size is in kilograms (kg) with lower and upper values given for the 95% confidence interval (CI).

		Spatial closure		**Temporal closure**		
Gear type	**Current overall**	**Restricted**	**Non-restricted**	**Oct-Nov**	**Rest of year**	**Both closures**
Traps						
Mean catch size (kg) per trip (CI)	1.41 (1.32-1.51)	1.84 (1.69-2.01)	1.14 (1.04-1.24)	1.71 (1.46-1.99)	1.36 (1.26-1.46)	1.11 (1.00-1.22)
Number of trips	1,284	576	708	215	1,069	604
Total weight of all catches (kg)	3,117	1,723	1,394	632	2,485	1,149
Income for mean catch per trip[a]	US$1.18	US$1.53	US$0.95	US$1.43	US$1.13	US$0.93
Gill nets						
Mean catch size (kg) per trip (CI)	1.53 (1.33-1.76)	1.40 (1.12-1.76)	1.64 (1.37-1.96)	2.62 (2.08-3.30)	1.14 (0.97-1.34)	1.21 (0.97-1.51)
Number of trips	319	137	182	114	205	118
Total weight of all catches (kg)	881	354	527	495	386	242
Income for mean catch per trip[b]	US$1.39	US$1.28	US$1.49	US$2.39	US$1.04	US$1.10
Both gear types						
Total trips	1,603	713	890	329	1,274	722
Total weight	3,998	2,077	1,921	1,127	2,871	1,391

[a] Median price paid for catches by trap fishers was US$0.83 per kilogram (see Chapter 4).
[b] Median price paid for catches by gill net fishers was US$0.91 per kilogram (see Chapter 4).

APPENDIX S6 – SUPPORTING INFORMATION FOR CHAPTER 6

Table S6.1. Characteristics of fishers and their fishing activity by age category. Standard errors (SE) are shown in parentheses. ANOVA results refer to differences between age categories for each characteristic (*df* = 4).

Characteristic	**Age category**					
	15-24	**25-34**	**35-44**	**45-54**	**55+**	**ANOVA**
Number of fishers in cluster (n = 151)	17	47	47	26	14	-
Mean catch per trip (kg)	1.42 (±0.37)	1.64 (±0.20)	2.08 (±0.21)	1.75 (±0.38)	0.97 (±0.19)	F=5.55 **p<0.001**
Mean proportion of catch sold per trip	55% (±8.1)	72% (±2.6)	68% (±2.1)	68% (±3.8)	53% (±4.6)	F=4.64 **p=0.001**
Mean effort (time spent fishing in hours) per trip	1.9 (±0.41)	1.7 (±0.10)	2.0 (±0.10)	1.8 (±0.18)	2.1 (±0.19)	F=2.73 **p=0.028**
Mean one way distance travelled (km)	5.0 (±0.48)	4.0 (±0.18)	3.9 (±0.12)	4.3 (±0.21)	3.0 (±0.16)	F=6.32 **p<0.0001**
Mean years of fishing experience	5.1 (±0.48)	9.4 (±0.25)	17.3 (±0.36)	25.6 (±0.98)	40.6 (±0.83)	F=418.19 **p<0.0001**
Mean number of people supported in household	2.3 (±0.34)	3.8 (±0.11)	4.9 (±0.07)	5.8 (±0.11)	6.0 (±0.20)	F=75.69 **p<0.0001**
Proportion with alternative livelihood	41%	70%	79%	88%	100%	-

APPENDIX S7 – SUPPORTING INFORMATION FOR CHAPTER 7

Appendix S7.1. Fisher responses by age category to a change in fishing intensity

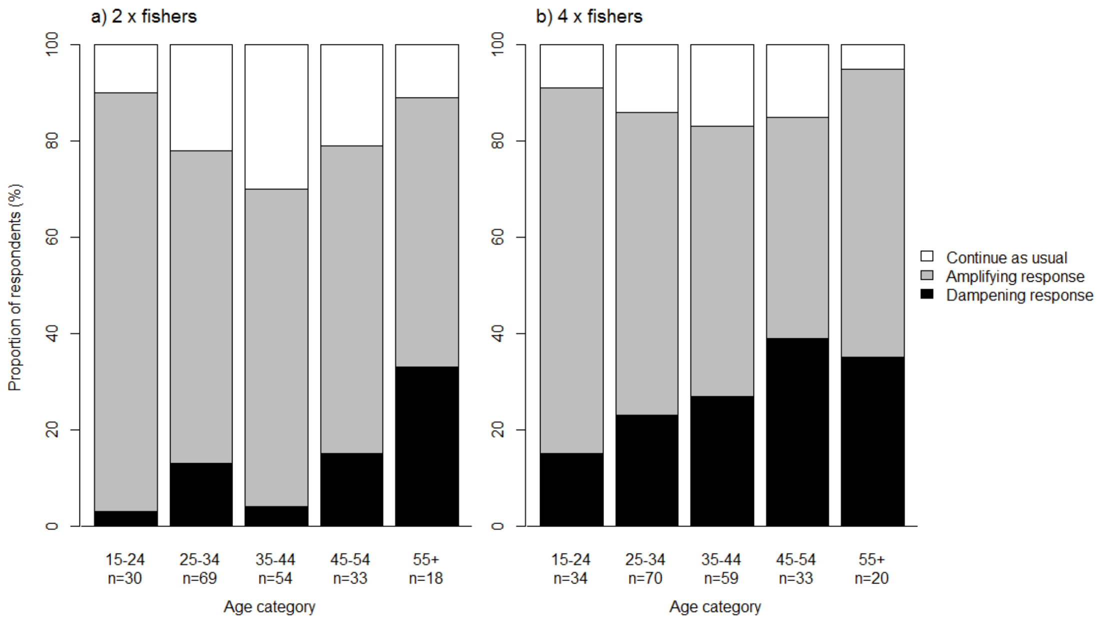


Figure S7.1.1. Fisher responses by age category to a hypothetical change in fishing intensity; a) a two-fold increase and b) a four-fold increase in the number of fishers using a fishing location.

Appendix S7.2. Adaptive responses to the frequency of good and bad days

Methods

During scenario interviews, fishers were asked how they would respond to changes in the number of good or bad fishing days as defined by the fisher; twice the number of good days as well as twice the number of bad days in both seasons. Using the approach described in Section 7.2.3.1.3 in Chapter 7, responses were categorised into the same three themes: 1) continuing as usual, 2) an amplifying adaptation, or 3) a dampening adaptation. Proportions of responses were not significantly different between seasons (Fisher's exact tests, $p > 0.25$) and therefore overall mean values were used.

Results

Of the 221 fishers interviewed, the majority (63%) stated that they would increase effort if there were twice the number of good fishing days, so as to catch even more fish, earn more money, or take advantage of good conditions (Figure S7.2.1a). Respondents that would increase effort would do so by adding equipment, spending more time fishing, or improving equipment. Just over one third (34%) stated that they would continue fishing as usual, including fishers from all age categories and using all types of gear. Some of these fishers (n = 10) stated that they would use the extra money earned from fishing to participate in other income-generating activities such as opening a shop, expanding rice cultivation, or trading rice to further increase their earning potential.

Although the majority of fishers (48%) stated an amplifying response to twice the number of bad fishing days, the range of adaptive responses were significantly different to those reported for twice the number of good fishing days (Fisher's exact test, $p < 0.0001$). Of the fishers who indicated an amplifying response, significantly more fishers would change gear or change location and significantly fewer fishers would increase effort (chi-square tests, $df = 1$, $p < 0.02$) (Figure S7.2.1b).

Fishers who stated they would change location in response to a two-fold increase in the number of bad fishing days indicated they would do so to try to catch more fish. Fishers stated they would switch to a location they were familiar with, that they perceived to be a good location with many fish, fewer fishers, and/or was open/spacious, or would alternate between locations as necessary. Of the fishers who specified an alternative location (n = 70), 42 (60%) nominated locations that were a median of 4.6km further from the village than their original location(s) and 28 fishers (40%) nominated locations that were a median of 4.6km closer to the village (Table S7.2.1). This indicates that on average, fishers who would usually fish closer to the village would shift their location by the same distance as those who would usually fish further from the village, and the two groups would effectively swap spatial distributions; the 'grass is greener' effect.

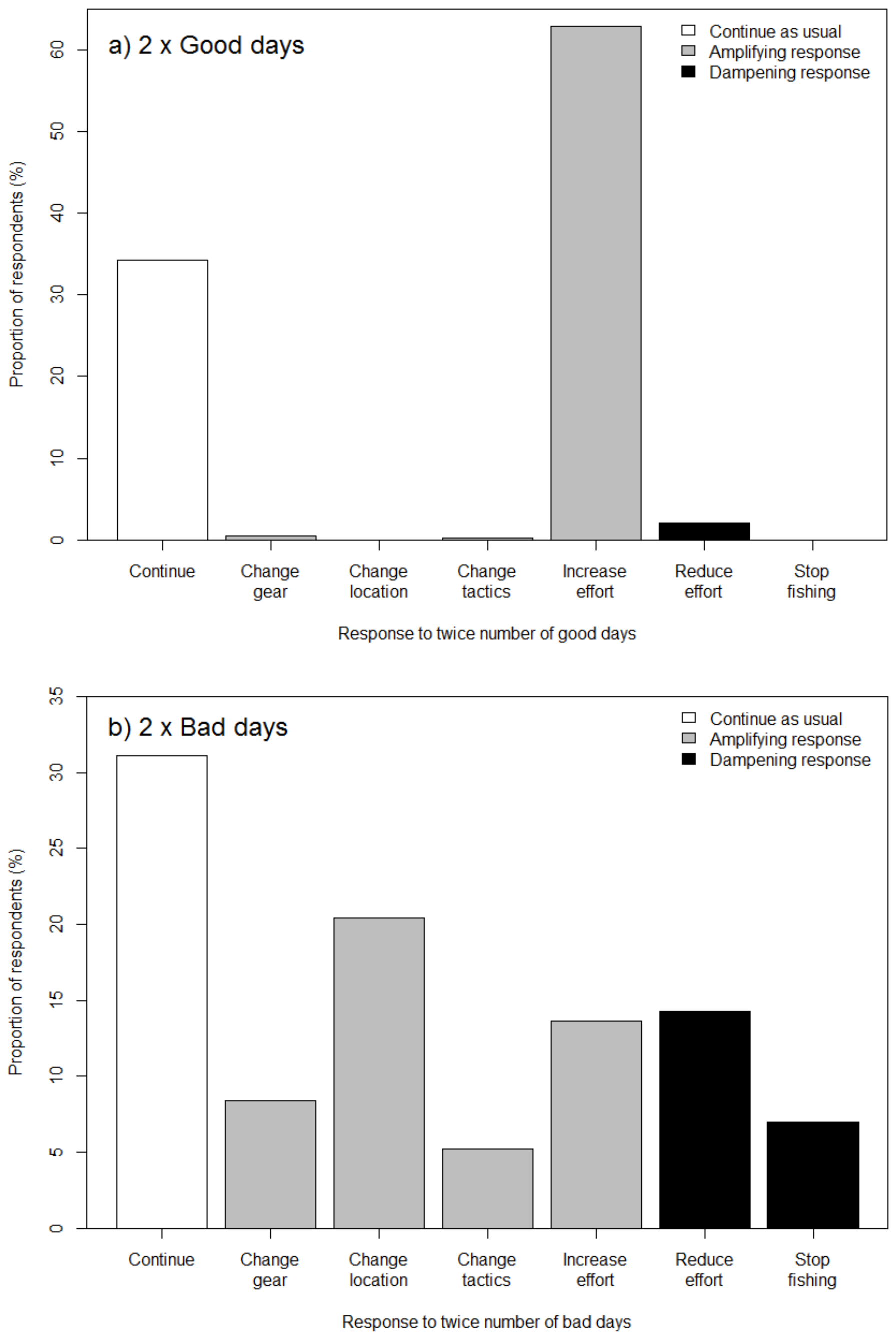


Figure S7.2.1. Fisher responses to a hypothetical change in the number of a) good fishing days and b) bad fishing days.

Table S7.2.1. Median distance travelled in kilometres to current and new fishing locations in response to a two-fold increase in the number of bad fishing days. Range is shown in parentheses.

Distance travelled	Fishers travelling further (n = 42)	Fishers moving closer (n = 28)
To current location(s)	6.0 (1.3 to 21.1)	12.1 (3.0 to 25.0)
To new location(s)	12.2 (2.2 to 27.9)	5.7 (0.1 to 20.4)
Change	4.6 (0.05 to 22.4)	-4.6 (-0.5 to -19.2)

Although the data indicate that most fishers aged up to 54 years would travel further from the village in response to twice as many bad fishing days, whereas most older fishers would move closer, travel responses did not differ significantly with fisher age (Fisher's exact test, $p = 0.311$). Travel responses also did not differ significantly with gear type (Fisher's exact test, $p = 0.257$).

Fishers who indicated that they would continue to fish as usual (31%) with a two-fold increase in the number of bad fishing days stated that they would do so because fishing was their primary livelihood, they needed food and/or money, there was no other feasible option, or they would simply make do while waiting for conditions to improve; responses were comparable across all fisher age categories. The majority of fishers who stated they would reduce their effort (n = 32) or stop fishing (n = 16), 77% and 68% respectively, stated they would work in rice cultivation as an alternative source of income.

Appendix S7.3. Adaptive responses to the removal of invasive aquatic plants

Methods

During background interviews, fishers were asked how best to manage the fishery. Some fishers indicated that the removal of invasive aquatic plants would improve water quality and re-open fishing locations. To examine the perceived impacts and adaptive responses to the removal of invasive aquatic plants as a management strategy, fishers were asked during scenario interviews how this action would affect them and how they would respond.

Results

Ninety-four percent of fishers interviewed (n = 221) stated their income from fishing would change if invasive aquatic plants (water hyacinth and giant salvinia) were removed from the lake, marsh, and channels; 98% of whom stated their income would increase – median anticipated increase was 100% (range 10% to 300%). Other benefits nominated by 210 fishers included: fish would be free to circulate (41%), marsh habitat and fishing locations would be renewed (32%), water quality and circulation would improve (10%), and access and travel in the marsh would improve (10%). Only 14 fishers (6%) stated that there would be costs or problems associated with removing invasive plants including statements such as 'it would be bad for the environment', 'plants are needed to conserve the humidity of the marsh and for rain', and 'plants provide a refuge for fish'.

Sixty-nine percent of fishers stated they would change location if invasive plants were removed; 65% of whom stated the reason for this change was because the new fishing location would be renewed and fish would move there. Other reasons included familiarity with the location, better quality of fish, larger fish, few fishers/thieves, and location characteristics (e.g., calm, sheltered from wind, shallow water). The majority of locations (73%) that fishers nominated were within marsh habitat; almost half (49%) of which are in the planned restricted area.

Of the 68 fishers who stated they would not change location, most (60%) stated the reason was because of familiarity with their current location(s). The remaining 40% stated that all fishing locations would be open if invasive aquatic plants were removed and fish would distribute widely, including into their current location(s), consequently eliminating the need to change.

APPENDIX S8 – SUPPORTING INFORMATION FOR CHAPTER 8

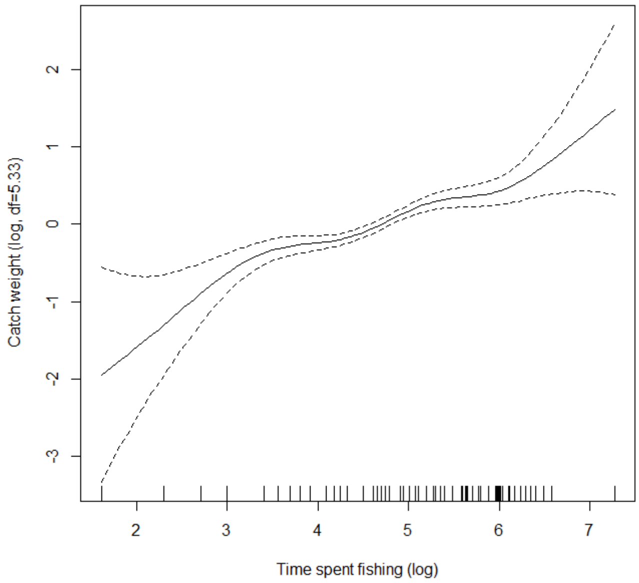


Figure S8.1. Prediction of catch weight from time spent fishing (solid line) with confidence intervals (dotted lines). Larger confidence intervals exist at the lower and upper end of time spent fishing.

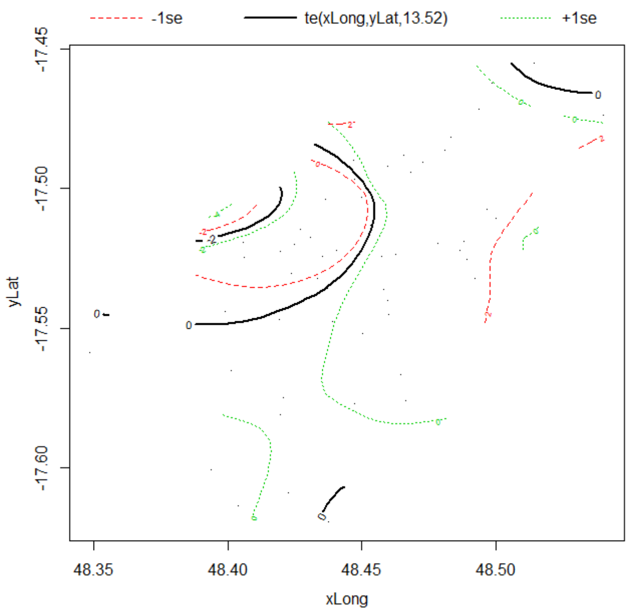


Figure S8.2. Prediction of catch weight from the longitude and latitude across the planning area (black lines) with confidence intervals (red and green lines) showing greater confidence in areas nearer the study village where more catch data is available.