

**UNITING HISTORIC PERSPECTIVES, HUMAN BEHAVIOUR, AND HABITAT  
USE TO ASSESS THE FUTURE FOR OVERFISHED SEAHORSES**

by

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## **Abstract**

This dissertation casts new light on reconciling fisheries and conservation. This reconciliation is particularly challenging for small-scale fisheries, because they are data-poor, but it is necessary to sustain biodiversity and food security. My research focuses on a typical small-scale fishery in the Central Philippines that catches seahorses.

I begin by providing an historic reconstruction of seahorse catch rates, the first of its kind despite a multi-decade, global trade for seahorses. I quantified fisher perceptions of the decline in trade to set historic baselines. Results showed that relying solely on recent decades of data could underestimate IUCN Red List extinction risk for historically exploited fish such as my study species, *Hippocampus comes*.

My research contributed robust approaches to incorporating fisher knowledge into quantitative assessments. I developed a novel approach to correct for inaccuracies in fishers' memories of past events and showed, for the first time, that conservation assessments are sensitive to assumptions made when analyzing local knowledge. Comparisons I made between catch rate trends estimated by resource users and those obtained using standard approaches (catch landings or underwater surveys) suggest that fisher interviews or logbooks are a reasonable proxy for more costly 'scientific' methods. In contrast, interviews do not allow inferences about the absolute value of catch rates at scales relevant to management/conservation.

My findings raise questions about the assumed sustainability of small-scale fisheries and identified new strategies to protect them. I conducted a novel analysis of spatial fishing

behaviour that illustrates the capacity of small-scale fisheries to exert considerable cumulative pressure on marine systems, despite their seemingly benign gears.

Finally, I conducted the first study on how depth refugia might help sustain coral-reef fish populations by combining fisheries modeling with IUCN Red List conservation assessments. Results emphasize the importance of locating some protected areas at depths inaccessible to small-scale fishers. Such tactics will contribute towards safeguarding biodiversity and food security in coral reef ecosystems.

In summary, my thesis takes valuable steps towards finding solutions for marine biodiversity and fisheries crises; it uses pragmatic approaches to deduce key information that is highly relevant to the futures of coral-reef fish and small-scale fisheries.

## **Preface**

Some of the chapters of this thesis and ideas therein are already published. A version of Chapter 2 has been published. O'Donnell, K.P., Pajaro, M.G. and Vincent, A.C.J. (2010) How does the accuracy of fisher knowledge affect seahorse conservation status? *Animal Conservation* 13: 526-533. Chapter 3 has been provisionally accepted at a peer reviewed journal, Chapter 4 is presently in review, Chapter 5 is in preparation for submission, and some of the ideas in Chapter 6 have been published. O'Donnell, K.P., Pajaro, M.G. and Vincent, A.C.J. (2010) Improving conservation and fishery assessments with local knowledge: future directions. *Animal Conservation* 13: 539-540.

I am the senior author on all papers and all have co-authors. I took primary responsibility for design, implementation, analysis, and writing of all chapters. Details of co-authorship contributions are outlined below.

Chapter 2 – Dr. Amanda Vincent came up with the original idea to use existing, unpublished interview and logbook data to reconstruct the history of seahorse fishing. I embraced the opportunity to turn this idea into reality and built on it by linking to IUCN Red List assessments and evaluating the effects of uncertainty on extinction risk. I also developed the methodology to evaluate the effects of uncertainty. I expanded on the original idea, developed and carried out all analyses, and drafted the text. My other co-author, Marivic Pajaro, along with Amanda Vincent, provided the data.

Chapter 3 – Dr. Amanda Vincent came up with the original idea to compare data-collection methods and provided fisher logbook and catch landings data. Co-author Dr. Phil Molloy provided guidance in learning R statistical software and also contributed edits,

comments and suggestions for additional analyses on numerous iterations of the paper. I collected all interview data, carried out all data management and analyses, and drafted the text.

Chapter 4 – I came up with the original idea to log fisher movements with GPS and depth loggers and to modify swept area method exploitation rate calculations to fit available data. I also devised the methodology for interpreting GPS and depth logger data. Co-authors Dr. Amanda Vincent and Dr. Jeremy Goldbogen contributed to the idea of this chapter through stimulating discussions and pointed comments on drafts of the manuscript. Analysis of diving behaviour was conducted by Jeremy Goldbogen and I conducted all other analyses. Logbook data were provided by Amanda Vincent. I collected all other information for the manuscript, carried out all data management, and drafted the text.

Chapter 5 – The original idea of evaluating evidence for a depth refuge from lantern fishing was mine, and co-author Dr. Phil Molloy contributed the idea of constructing a model to evaluate the possible contribution of the refuge to shallow areas and provided edits and comments on numerous iterations of the paper. Both Phil Molloy and co-author Jon Schnute contributed to the methodology of the paper by providing guidance on constructing and validating the model. I collected all information for the manuscript, carried out all data management and analyses, and drafted the text.

In addition to contributions described above, Dr. Amanda Vincent provided edits, comments and suggestions for improvement on drafts of each chapter.

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## **Chapter 1: Introduction**



## **Rationale**

The capacity of humans to cause declines in wildlife species, especially from direct exploitation, is immense and growing. On land, hunting has caused extinction waves of megafauna (Alroy, 2001). In the sea, fishing has driven whales, manatees, dugongs, seacows, monk seals, sharks, rays and other large marine vertebrates to be functionally or entirely extinct in most coastal systems (Bertram & Bertram, 1973; Kenyon, 1977; Casey & Myers, 1998; Clapham, Young & Brownell Jr, 1999; Baum *et al.*, 2003; Lotze & Worm, 2009). The largest predatory marine fishes have also largely succumbed to extensive, global declines due to chronic overfishing (Pauly *et al.*, 1998; Jackson *et al.*, 2001; Myers & Worm, 2003). It is generally agreed that conventional approaches to control fishing have failed to sustain fish populations or fisheries (Pitcher, Hart & Pauly, 1998; Pauly *et al.*, 2002). Given the importance of fish to human food security (Delgado, 2003; Alder & Sumaila, 2004) and their roles as part of productive and functioning ocean systems (Field *et al.*, 1998; Worm *et al.*, 2006) action must be taken to prevent further declines and to restore the abundance of sea life.

This dissertation addresses the major challenge of applying pragmatic approaches to assess and recover fish species threatened by overfishing. Getting this right for small-scale fisheries will be important for sustaining biodiversity and food security, and particularly challenging. I shall describe in more detail the challenges facing the world's fisheries and fish conservation efforts. I shall then outline emerging approaches to reconcile the two paradigms, and describe metrics to measure progress toward healthy fish populations and fisheries.

## **Fisheries**

Fisheries are globally important and in trouble. Fishing directly employs at least 51 million people (Berkes *et al.*, 2001), and international trade in fisheries products generates more foreign exchange than the revenues earned from any other traded food commodity such as rice, cocoa, coffee or tea (FAO, 2003). The tremendous growth of fisheries in the 20<sup>th</sup> century reached a plateau by the 1980s (FAO, 2009) as new areas to exploit were exhausted (Swartz *et al.*, 2010). The numerous examples of overexploitation causing population declines (Myers & Worm, 2003; Hutchings & Reynolds, 2004) and alteration of ecosystems (Pauly *et al.*, 1998; Watling & Norse, 1998; Jackson *et al.*, 2001; Worm *et al.*, 2006), have led many to perceive a global fisheries crisis (Buckworth, 1998). Collapses of iconic species like Atlantic Cod ended centuries-old fishing traditions in New England and eastern Canada (Myers, Hutchings & Barrowman, 1997). There are examples of fishes recovering from overfishing (e.g. Worm *et al.*, 2009), but in many cases recovery has not occurred, even after fishing ceased (Hutchings, 2000; Hutchings & Reynolds, 2004).

While international attention to global fisheries problems has focused on industrial, mostly temperate, single-species fisheries, there is growing recognition that small-scale, multi-species fisheries, specifically those on coral reefs, need substantially more attention (Mahon, 1997; Bellwood *et al.*, 2004; Castilla & Defeo, 2005). The distinction between industrial and small-scale fisheries tends to be by boat size, leading to spatial segregation: industrial fishing refers to large vessels that can operate in the open-ocean, while small-scale fishing refers to small, coastal boats (Chuenpagdee *et al.*, 2006). One reason that attention has turned to small-scale fisheries is because they employ the vast majority of the world's

fishers. It is estimated that fifty million of the world's fifty-one million fishers are employed in small-scale fisheries (Berkes *et al.*, 2001), though there is some debate about the exact number of people employed (reviewed in Béné, 2006). Small-scale fisheries occur globally, but are of particular concern in developing countries which overlap with coral reef systems (Pauly *et al.*, 2002; Burke *et al.*, 2011) that have been broadly degraded by overfishing (Jackson *et al.*, 2001; Pandolfi *et al.*, 2003). In addition, small-scale fisheries are poorly studied and rarely managed (Berkes *et al.*, 2001; Parma *et al.*, 2003; Chuenpagdee *et al.*, 2006; McClanahan *et al.*, 2009).

Although they may catch half the world's fish, basic information needed to assess the effects of small-scale fishing is largely unavailable - contributing to the challenge of management. Most countries' official statistics do not specify catch from small-scale fisheries, thus implying that such catch is negligible. In fact, though, the few studies that estimated catch suggest that the few official statistics of small-scale catches are gross underestimates (Zeller *et al.*, 2006a; Zeller *et al.*, 2007; Jacquet *et al.*, 2010). This under-reporting tends to under-estimate the effects small-scale fishing on marine systems and their contribution to livelihoods (Zeller, Booth & Pauly, 2006b). Correcting for under-reporting suggests that globally small-scale fishers catch roughly the same amount (about 30 million tonnes year<sup>-1</sup>) as industrial fishers (Chuenpagdee *et al.*, 2006; Pauly, 2006).

In addition to poor documentation of small-scale catches, there is often no available biological information, even for the economically important species (Sadovy, 2005). Fish biological data are especially scarce on coral reefs because vast areas of tropical reefs have not been studied (Simon & Polunin, 1996). Data poverty has contributed to small-scale fisheries being poorly managed, if at all (Munro, 1996; Berkes *et al.*, 2001). Most are open-

access fisheries and in many there is almost no economic constraint on effort; the possibility of a decent catch, however remote, is sufficient incentive for unremitting exploitation of resources (Munro, 1996).

Small-scale fisheries have historically been considered ‘unmanageable’ (Orensanz *et al.*, 2005) because conventional fisheries science and management lack methods to address the complexities of small-scale fisheries. While industrial fisheries specialize primarily on a single gear that catches a single species that is landed at few commercial ports, small-scale fisheries employ many gears catching many species with small landings in many places (Russ, 1991). This complexity renders it impractical to collect typical data on fishery status (e.g. catch rates) and implement command-and-control measures (e.g. quotas) for small-scale fisheries. Even if catch and effort data were available for small-scale fisheries, conventional assessment models (e.g. Stock Reconstruction Analysis; Walters, Martell & Korman, 2005) assume that fish populations are mobile, evenly distributed, and that large enough areas are considered to nullify effects of movement. These assumptions do not hold for small-scale fisheries (Parma *et al.*, 2003) such as those on coral reefs where fish are relatively stationary, spatially heterogeneous, structured as metapopulations, and where movement may play an important role in population persistence (Gotelli, 2001). Overall, stock assessment-based fishery management has been too impractical, expensive, incomplete, and uncertain to address the needs of small-scale fisheries (Mahon, 1997; Berkes, 2003).

Developing practical approaches to manage small-scale fishing pressure will be especially important for conserving the wealth of biodiversity in coral reefs. Small-scale fisheries tend to co-occur with some of the most biodiverse habitats on earth, coral reefs (Connell, 1978; Roberts *et al.*, 2002). Although coral reefs account for only 0.1% of the

world's ocean, their fisheries resources provide tens of millions of people with food and livelihood (Moberg & Folke, 1999; Pauly *et al.*, 2002). Traditionally, reef fisheries provided tropical coastal communities with an important, sometimes sole, source of protein and livelihood (Sadovy, 2005). However, as early as 1900, these fisheries caused substantial degradation to reef systems (Jackson *et al.*, 2001; Pandolfi *et al.*, 2003). This degradation has been magnified more recently with increased demand for reef species on the international market, e.g. for the live reef food-fish trade (Sadovy, 2005). With both local and international pressures, fishing continues to be among the most pervasive threats to coral reefs (Burke *et al.*, 2011), especially in Southeast Asia (Newton *et al.*, 2007). Increases in reef fishing pressure are likely because globally 75% of coral reefs occur in developing countries where human populations are still increasing (Pauly *et al.*, 2002).

### **Fish conservation**

Fishes are heavily exploited wild animals, but have not been a conservation priority. Instead, global conservation efforts have focused on charismatic, largely terrestrial, megafauna (Clark & May, 2002). Among vertebrates, fishes are the least well-assessed for extinction risk on the IUCN Red List ([www.iucnredlist.org](http://www.iucnredlist.org)) - the most universally accepted and comprehensive system for assessing species extinction risk (Colyvan *et al.*, 1999; de Grammont & Cuarón, 2006; Rodrigues *et al.*, 2006b). Just 11% of described fish species are assessed (as compared to 100% of mammals and birds; Vié, Hilton-Taylor & Stuart, 2009; Zamin *et al.*, 2010).

Perhaps fishes have not received conservation attention because they have been assumed to be more resilient to exploitation than terrestrial species. Fish have been hunted for food for millennia (Jackson *et al.*, 2001; Pauly, Watson & Alder, 2005) and were once bountiful which likely created assumptions that as a resource they were inexhaustible (Hutchings, 2001 and citations therein). As recently as 1996, when the first commercially exploited marine fish were listed on the IUCN Red List ([www.redlist.org](http://www.redlist.org)), fisheries experts argued that fishing could not drive fish to extinction (Vincent & Hall, 1996; Mace & Hudson, 1999). Red Listing aims to “convey the urgency and scale of conservation problems to the public and policy makers, and to motivate the global community to try to reduce species extinctions” (IUCN, 2008a). It achieves this by assessing species (e.g. based on population change) and then assigning them a threatened status, from least concern all the way to extinct. Despite the rigor of the Red Listing process, fisheries experts argued that higher fecundities in marine fishes and large scale dispersal would always allow recovery from low numbers and that high natural variability in population size rendered Red List decline criteria too conservative for marine fishes (Matsuda, Yahara & Uozumi, 1997; Musick, 1999). In response, Red List Criterion A that addresses population declines was revised to address their concerns (Dulvy, Sadovy & Reynolds, 2003). A comparison of these revised Red list decline criteria with fisheries stock assessments suggests that both methods provide consistent warnings of population collapse (Dulvy *et al.*, 2005).

Current evidence suggests that, in fact, fishes are just as vulnerable to extinction as terrestrial wildlife. Maximum reproductive rates (recruits per spawner/adult) for fishes are similar to those estimated for terrestrial mammals such as the grizzly bear (*Ursus arctos*), bighorn sheep (*Ovis Canadensis*) and white-tailed deer (*Odocoileus virginianus*)

(Hutchings, 2001). Across fish species that vary widely in fecundity (number of gametes), reproductive rates are fairly consistent (Myers, Brown & Barrowman, 1999). Genetic evidence of juvenile fish returning to natal reefs, suggests that population distributions and sizes may be considerably more restricted than is assumed (reviewed by Dulvy *et al.*, 2003). In addition, fish populations fluctuate no more from year to year than do populations of mammals and butterflies (Fagan *et al.*, 2001). Therefore, there is little evidence to suggest that fishes warrant different at-risk criteria than other organisms (Hutchings, 2000; Dulvy *et al.*, 2003; Dulvy *et al.*, 2005). In fact, fishes may be more vulnerable to extinction than taxa like butterflies and birds (Fagan *et al.*, 2001).

Although efforts to quantify the global conservation status of marine fishes are in their infancy, continued pressures are cause for concern. Although only 32 populations and species of marine fish have been documented as experiencing local, regional, or global extinctions, our state of knowledge is so poor that many more extinctions may well have occurred without being noticed (Dulvy *et al.*, 2003). Groupers (Serranidae) are the first fish family for which all species' risk of extinction has been assessed using IUCN Red List criteria (IUCN, 2001). For this family, at least 12% of the world's 161 species have been listed in threatened categories (Critically Endangered, Endangered, or Vulnerable), another 14% are Near Threatened, and 30% are considered to be Data Deficient (Polidoro *et al.*, 2008). Similarly, the first regional assessment of all fish species in the Mediterranean showed that while only 8% of the 519 native fishes assessed were classified as threatened, about one-third were classified as Data Deficient. The good news was that more than half of the assessed species were not facing immediate risk of extinction (Least Concern). On the

other hand, with over half of species subject to fishing, and many more not protected or effectively managed, the true proportion of threatened species might be much higher.

### **Reconciling small-scale fisheries and conservation**

There is a call for a pragmatic and effective system to maintain long-term viability of fish populations, benefiting both fisheries and conservation. This reconciliation of fisheries and conservation would contribute to sustainable fisheries - those that meet the needs of the present without compromising the ability of future generations to meet their own needs (UN, 1987). The global fishery crisis is a product of the conventional fisheries science and management paradigm (Buckworth, 1998). This conventional system relies on expert scientific knowledge to make top-down decisions, aims for the control of nature, seeks to maximize yield from wild species, and treats people as if they were separate from the environment (Pitcher *et al.*, 1998; Berkes, 2003). Although entire books are devoted to ‘re-inventing fisheries management’ (e.g. Pitcher *et al.*, 1998; Pomeroy & Andrew, 2011), problems persist. I here outline five promising approaches and four indicators that can measure progress toward healthy fish populations in small-scale fisheries.

### **Approaches**

Instead of the conventional approach to fishery assessment and management, a more pragmatic system would take five approaches, each of which I assess in turn:



- 1) tap into unconventional data-sources (Walters, 1998; Berkes, 2003) for rapid, cost-effective, and reliable information on species abundance and exploitation;
- 2) consider a broader social-ecological system (Berkes, Folke & Colding, 2000) including the behaviour of fishers (Hilborn, 1985; Salas & Gaertner, 2004);
- 3) evaluate the effects of uncertainty on assessments and management options (Ludwig, Hilborn & Walters, 1993; Walters, 1998; Kinzig & Starrett, 2003);
- 4) reach conclusions and recommendations for action despite incomplete knowledge (Ludwig *et al.*, 1993; Johannes, 1998; Robinson, 2006); and
- 5) err on the side of precaution – aiming to protect the viability of the resource (Ludwig *et al.*, 1993).

### ***Tap into unconventional data sources - local knowledge***

Despite a growing interest in incorporating local knowledge into quantitative conservation (Danielsen, Burgess & Balmford, 2005) and fishery assessments (Neis *et al.*, 1999b; Johannes, Freeman & Hamilton, 2000), there remain practical impediments to its use. Local knowledge can be defined as practical knowledge that has not necessarily been passed down through generations like traditional ecological knowledge (Berkes, 2003). Documenting the knowledge of resource users can be an inexpensive (Moore *et al.*, 2010; Rist *et al.*, 2010) and effective way of gathering data on species, catches, ecology, and habitat, especially to understand changes over time (Johannes, 1981; Mackinson & Nottestad, 1998; Neis *et al.*, 1999b; Johannes *et al.*, 2000; Scholz *et al.*, 2004). Interviews with resource users (e.g. Sáenz-Arroyo *et al.*, 2005) and their daily written logs (Baum *et al.*, 2003;

Rosenberg *et al.*, 2005) have even uncovered population declines that were previously undetected by western science (e.g. Gilchrist, Mallory & Merkel, 2005).

Since the first studies that demonstrated the utility of local knowledge to quantitative fishery management (e.g. Neis *et al.*, 1999b), there have been few conservation or fishery assessments based on local knowledge (Brook & McLachlan, 2008). Perhaps the most significant impediment to more widespread incorporation of local knowledge into quantitative assessments lies in appraising its reliability (in terms of accuracy and precision), especially because there are no formal guidelines for interpretation or analysis of local knowledge data. In addition, while biases in local knowledge have been acknowledged (Neis *et al.*, 1999b; Ainsworth, Pitcher & Rotinsulu, 2008; Daw, 2008a), there is little understanding about how different assumptions of bias affect quantitative conservation and fishery assessments.

### ***Understand fisher behaviour***

Understanding the spatial dynamics of fishing behaviour can be a pragmatic way to improve assessment and management of industrial and small-scale fishing alike (Hutchings & Myers, 1994; Booth, 2000; Wilen *et al.*, 2002). There is an increasing awareness that knowledge of the behaviour and decision-making process of fishers is essential for effective assessment and management (Hilborn, 1985; Salas & Gaertner, 2004; Walters & Martell, 2004; Branch *et al.*, 2006). For example, knowledge of spatial fishing patterns allows for better interpretation of catch rates (catch per unit of effort – CPUE) because fish population declines can be masked by changes in spatial distribution of fish and/or fishing (Gillis,

Peterman & Tyler, 1993). Such misunderstandings may help explain the causes of the most severe fishery collapses (e.g. Atlantic cod, Peruvian anchovy; Hilborn & Walters, 1992).

Although fisher behaviour affects trends in catch rates, little is known about fishing behaviour, especially for small-scale fisheries (Salas & Gaertner, 2004).

Understanding where fishers fish also highlights areas where fishing effort is limited or non-existent (natural refugia), areas that could be important for fishery sustainability (Pauly *et al.*, 2005). Low-tech, cost effective interviews with small-scale fishers about where they fish provides useful, but coarse, information (e.g. Daw, 2008b). Digital tracking technologies, more commonly applied to monitor animal movements (e.g. digital tags; Goldbogen *et al.*, 2008) or industrial fisheries (e.g. GPS tracking; Bertrand *et al.*, 2007) are becoming more affordable and demand less of fishers' time than interviews. These higher-tech, spatial, data-collection methods could provide complementary detailed spatial information to interviews.

### ***Evaluate effects of uncertainty on assessments and management options***

The complex and dynamic nature of human interactions with ecosystems means that scientists and decision-makers must expect and learn to manage associated uncertainty (Ludwig *et al.*, 1993; Myers, 1995; Kinzig, 2003; Robinson, 2006). Many important processes that structure fish populations, and therefore affect their resilience to fishing (e.g. population connectivity through dispersal, Jones *et al.*, 2009) are still poorly understood (Sale *et al.*, 2005). When urgent decisions need to be made about threatened species, modeling is one pragmatic tool that can combine available data from various sources and

systematically incorporate uncertainties to weigh management or recovery options. As a representation of how we think the world works, a model can allow for assumptions to be explicitly stated, uncertainties quantified, data gaps identified, and future data gathering efforts prioritized (Hilborn & Walters, 1992). Although classic fisheries modeling approaches are technically complex and relatively data-hungry, even simple or conceptual models can be useful (Milner-Gulland & Rowcliffe, 2007). Furthermore, it is becoming possible to populate models with data from unconventional sources (e.g. Ainsworth *et al.*, 2008).

### ***Reach conclusions and recommendations for action despite incomplete knowledge***

Reaching conclusions in the absence of complete knowledge is necessary because conservation problems are often too complex for scientific methods to provide robust answers (Robinson, 2006). Problems such as conservation of world fish species are ‘wicked problems’ (Rittel & Webber, 1973) that involve many disciplines and have no test for a solution (Ludwig, Mangel & Haddad, 2001). Because of the complexity of these problems, controlled and replicated experiments are not possible in large-scale, marine systems such that optimum (maximize catch without threatening species viability) levels of exploitation can probably only be determined by trial and error (Walters, 1986; Kinzig & Starrett, 2003). A result is that consensus among scientists is unlikely, to a point where some have recommended against consensus as a requirement for action (Ludwig *et al.*, 1993). Careful consideration of the management objectives and controls that are practical for threatened

marine resources for which few or no data are, or will be, available is overdue (Johannes, 1998). As Johannes so eloquently put it,

“The key question should not be ‘what data are needed to make sound management decisions?’ but rather, ‘what are the best management decisions to make when such data are unattainable?’” (Johannes, 1998).

### ***Err on the side of precaution – aiming to protect the viability of the resource***

Uncertainty mandates a more cautious approach to resource exploitation than is the norm (Ludwig *et al.*, 1993). In reaction to uncertainty, fisheries stakeholders often prefer that governments prove negative effects of fishing before introducing more restrictive fishing regulations (Walters & Martell, 2004), but a number of international agreements pertaining to fisheries reject this notion. Both the *Law of the Sea* (UN, 1995) and the *FAO Code of Conduct for Responsible Fisheries* (FAO, 1995) reverse the ‘burden of proof’ in favor of the precautionary principle. According to this principle, management systems should err on the side of conservation, especially when there is chance of irreversible changes that may degrade the equity of future generations (Berkes *et al.*, 2001). Erring on the side of conservation means that the ‘burden of proof’ shifts such that the onus is on fisheries stakeholders to show that odds of harm due to fishing are low (Charles, 1998).

### **Indicators of change**

To understand the efficacy of the five approaches I outline, measurable indicators are required to track progress toward healthy fisheries and fish populations. These indicators are

also helpful because typical, more exhaustive metrics (e.g. direct counts of fish population size and changes in response to fishing) are rarely available for small-scale fisheries and threatened species. While some of the four indicators described below are commonly used in more data-rich situations, I will either use more pragmatic data to calculate them or revise the indicator itself to suit my limited data scenario.

### ***Trend in CPUE***

Trends in CPUE have been a key indicator of the effects of fishing on fish (despite flaws; Harley, Myers & Dunn, 2001), but tend to be collected by well-funded governments and are therefore largely unavailable for small-scale fisheries (FAO, 2009). I explore a more local, rapid approach to collecting CPUE data by using fisher-derived methods such as logbooks and interviews. In a short period of time, interviews can provide an historic baseline for fish abundance and fishing pressure, but little research had previously been done to examine the reliability (precision and accuracy) of collecting CPUE from fisher perceptions.

### ***Extinction risk – IUCN Red Listing***

Percent decline in CPUE, classically a fisheries metric, can also be used to evaluate species conservation status using IUCN Red List criteria. Red List assessments are most commonly based on ecological data (McClenachan, Ferretti & Baum, in review), but

exploring results of assessments based on alternative, more pragmatic sources like local knowledge could speed the assessment process for under-assessed taxa such as fishes.

### ***Exploitation rate***

The proportion of a population removed by fishing, exploitation rate, is an important indicator to assess overfishing. More specifically, exploitation rate quantifies the mortality associated with fishing relative to some pre-determined reference exploitation rate.

However, typically it is too data hungry to be practical for use in small-scale fisheries. As a common fishery reference point, there are general guidelines for safe ranges of exploitation, based on life history. For example, it has been suggested that maintaining a fixed exploitation rate at a relatively small fraction (0.5-0.8) of the natural mortality rate will provide nearly optimum yields (Martell, Walters & Wallace, 2000). Although exploitation rate is usually calculated as the proportion of a population caught, population size is often not documented in data-poor small-scale fisheries (Orensanz *et al.*, 2005). As an alternative, fishing mortality can be calculated as the proportion of a population's distribution that is swept by fishing effort during a year (the 'swept area' method, Walters & Martell 2004). This fishing mortality ( $F$ ) can be converted to exploitation rate ( $U$ ) as  $U = 1 - e^{-F}$ . If data on species distribution are unavailable, fisher interviews could fill the gap, and the area swept by fishing can be gathered relatively quickly and inexpensively using GPS technology as described above.

### ***Area protected from fishing***

The presence of either natural or managed refugia can be an important indicator of extinction risk and fishery sustainability. Natural refugia alone can promote sustainability because extinction from overexploitation is less likely if some of its range remains untouched (Clayton, Keeling & Milner-Gulland, 1997; Pauly *et al.*, 2005). Natural refugia in the marine realm are disappearing (Morato *et al.*, 2006; Swartz *et al.*, 2010) and little is understood about how fished populations are linked to those in refugia (Tyler *et al.*, 2009). Exploring fisher behaviour can help identify potential refugia by outlining constraints on fishing (inaccessible habitats, depths, etc). Conversely, if fishing behaviour changes in a way that expands the reach of exploitation into former refugia, then populations and fisheries may quickly become threatened. Managed refugia can be used to compensate for a lack of knowledge about the fishery, by taking the precaution of excluding the fishery from accessing a sufficiently large proportion of the resource (Mahon, 1997). The location and size of refugia required to protect species and maintain fisheries are still debated (reviewed by Gaines *et al.*, 2010).

### **Context and collaborations**

I have found a system that allows me to explore practical approaches to assessing and addressing a threatened fish exploited by a small-scale fishery. My research is focused on seahorses caught by lantern fishers in the Danajon Bank region of the Central Philippines.



Because of concerns over un-sustainable international trade in seahorses for use in traditional medicines, curiosities, and aquarium display, all species of seahorse (genus *Hippocampus*) were listed on Appendix II of CITES in 2002 (Foster & Vincent, 2005). This listing does not ban the trade of seahorses, but instead CITES member countries must prove any seahorse exports are not detrimental to survival of their wild populations. One difficulty is that we lack understanding of the status of most seahorse populations (25 of the 33 IUCN Red List assessed species are listed as Data Deficient). Many seahorses are caught in small-scale fisheries, which are globally poorly understood and ineffectively managed (Berkes *et al.*, 2001; Parma *et al.*, 2003; Chuenpagdee *et al.*, 2006; McClanahan *et al.*, 2009).

Devising practical ways to assess and manage small-scale fisheries in the Philippines is a pressing issue. Reliance on fish for food (Fernandez, 2009), in combination with high human population pressure (human population growth rates have grown on average 5.4% per year from 1900 to 1977; Pauly, 2000) have resulted in intense pressure on marine resources in the Philippines. The vast majority of people (90%) who fish in the Philippines are engaged in small-scale fisheries (Luna *et al.*, 2004). On paper, the country has laudable management goals (e.g. manage for MSY, Fernandez, 2009) and progressive policies (e.g. management of marine resources has been devolved to local governments) and certainly there have been some successes (e.g. establishment and successful recovery of fish and fishery resources at Apo island; Russ *et al.*, 2004), but poor implementation of these rules helps explain the widespread and severe habitat degradation and collapse of fishery resources (Pauly, 2000; Green *et al.*, 2003; Stobutzki *et al.*, 2006). Historically the Philippines was a major exporter of seahorses (Vincent, 1996), but exports have officially ceased since the domestic Fisheries Code interpreted the CITES listing of seahorses as a complete ban on

extraction, despite the intended sustainable use provisions of CITES Appendix II (Morgan & Lourie, 2006).

The Danajon Bank region of the Central Philippines is of particular conservation concern because high biodiversity, destructive fishing, and poverty are magnified here. Danajon Bank is a 145 km long double barrier reef that hugs the northern and western coasts of the province of Bohol and has an area in excess of 2350 km<sup>2</sup> (Hansen *et al.*, in press). This region has been shown to represent the peak of global marine biodiversity (Carpenter & Springer, 2005) and to contain the greatest diversity of seahorses (Lourie *et al.*, 1999). Destructive fishing (e.g. trawling and blast fishing) has degraded a once-rich coral reef system to the point where marine life is grossly depleted and coastal communities are among the poorest in the Philippines (Green *et al.*, 2003). Overfishing may also reach global peaks here (Lachica-Aliño, Wolff & David, 2006), raising conservation and food security concerns (Roberts *et al.*, 2002).

The lantern fishery in the Danajon Bank region, whose fishers are locally known as manuga'ay, is a typical small-scale fishery that is open access, spatially-dispersed, catches many species, and has never been formally assessed for sustainability. Lantern fishers sling a kerosene lantern across the bow of a small outrigger canoe at night to illuminate the reef below while they swim and free dive in search of fish, including seahorses, and invertebrates.

Assessment of lantern fishing effects on seahorse populations of this region began in 1995 with the establishment of a community-based research and management programme (Martin-Smith *et al.*, 2004; Vincent *et al.*, 2007; O'Donnell, Pajaro & Vincent, 2010) by Project Seahorse. One focus of research has been on understanding the life history, ecology, and vulnerability of the most exploited seahorse species, the tiger tail seahorse, *Hippocampus*

*comes* Cantor 1850 (Perante, Vincent & Pajaro, 1998; Morgan & Lourie, 2006; Morgan & Vincent, 2007). As part of a larger effort to estimate the extent of the global trade in seahorses (Vincent, 1996), documentation of seahorse landings on Danajon Bank revealed very low CPUE (Vincent *et al.*, 2007), especially compared with historical accounts (Pajaro *et al.*, 1997). In addition, qualitative measures suggest that the fishery is unsustainable (Martin-Smith *et al.*, 2004).

Lantern fishers and their communities have taken the first steps to address declining seahorse catches. Consultations with fishers, managers, and scientists about preferred methods of addressing declines revealed that MPAs and Minimum Size Limits (MSLs) are the most preferred management options (Martin-Smith *et al.*, 2004). Many of the communities of the Danajon Bank region have embraced community-based management of no-take marine protected areas (MPAs) as a remedial fishery management approach (Hansen *et al.*, in press). While one community perceived their MPA to be improving fish populations and catches (Yasué, Kaufman & Vincent, 2010), the effect of MPAs on seahorses remains unclear and MPAs alone will certainly not be enough to recover depleted populations and catches. The first prediction of the effects of management options on *H. comes* viability suggested that slot sizes (smallest and largest individuals are not fished) would be the most precautionary tool to hedge against fishing driving *H. comes* extinct (Morgan, 2007).

Although the lantern fishery is now arguably the best studied fishery for seahorses in the world, robust assessments of the effects of fishing on populations are required to inform prioritization of recovery/management options. The need to assess and address seahorse exploitation is especially urgent because fishing effort continues unchecked. Despite the

national ban on extraction of seahorses even for local use, fishing continues in some areas due to lack of enforcement of the ban (O'Donnell personal observation) and a persistent need for fishers to maintain a livelihood.

To explore pragmatic ways of quantifying extinction risk and lantern fishery sustainability on seahorses, I collaborated closely with the Project Seahorse Foundation for Marine Conservation (PSF), a Filipino NGO. It has a strong tradition of facilitating community-based management of marine resources in Danajon Bank and its ties with communities afforded me the opportunity to live in the fishing villages where I conducted my research. My assessments of the characteristics of the lantern fishery, its effects on seahorses, and what results mean for the future can contribute to PSF's ongoing efforts to assist communities in protecting seahorse populations and improving catches. In addition, my work can assist PSF in its advisory role to the Philippines federal fisheries agency that imposed the ban on seahorse fishing.

This study system allows me to explore the five pragmatic approaches and four indicators of success I described earlier to evaluate the viability of fish populations and fisheries.

### **Research questions**

My research applies unconventional approaches and indicators to assess risk of extinction and fishery sustainability. I also evaluate the advantages and limitations of the approaches and indicators, and use them to explore assumptions about the sustainability of small-scale fishing. Despite the uncertainty that will always accompany studies of such

complex systems, I strive to make reasonable recommendations to those responsible for ensuring the sustainability of populations and fisheries.

The following five overarching questions map onto the pragmatic approaches and indicators described above:

- 1) How has CPUE changed over time according to fisher perception and what does that tell us about extinction risk? (*approach*: novel data; *indicators*: CPUE, Red Listing)
- 2) How do assumptions about the accuracy of fisher recall affect the assessment of extinction risk? (*approach*: novel data, consider uncertainty; *indicator*: Red Listing)
- 3) How comparable are three methods of quantifying CPUE (fisher interviews, fisher logbooks and ‘scientific’ catch landings)? (*approaches*: novel data, consider uncertainty; *indicator*: CPUE)
- 4) What are the spatial dynamics of small-scale fishing? And what does an estimate of exploitation rate say about the sustainability of small-scale fishing? (*approach*: consider fisher behaviour; *indicator*: exploitation rate)
- 5) What role does a depth refuge play in preventing short-term risk of species extinction and in supporting shallow catches given current exploitation rates? (*approach*: novel data, consider uncertainty, be precautionary; *indicator*: Red List, area protected)

I explore these questions in the data chapters and a general discussion described in more detail below.

## **Thesis outline**

This thesis has four data-based research chapters, followed by a general synthesis and discussion of implications of the work for conservation and management.

In Chapter 2, I link fisher interviews and fisher logbooks to reconstruct the history of CPUE as perceived by fishers. I also quantify assumptions about how accurately fishers report past events (retrospective bias) and examine how the assumption I make about retrospective bias affects the characterization of changes in the fishery and extinction risk.

In Chapter 3, I delve deeper into the reliability of data elicited from resource users, comparing them with data obtained using scientific approaches. I evaluate the degree of consensus among three methods of quantifying catch rates (CPUE): fisher interviews, logbooks, and catch landings. I compare trends and absolute values of CPUE among data-collection methods for the lantern fishery. I also test whether interview reports are correlated with recorded landings to evaluate how interchangeable the two methods might be.

In Chapter 4, I quantify spatial fishing patterns and depth limits to lantern fishing, using GPS and depth tags. These data, superimposed onto satellite images of habitat and combined with direct observation and interviews, create a three-dimensional representation of spatial fishing behaviour that allows for estimation of exploitation rate – an important indicator of sustainability.

Given steep declines in catch rate (Chapter 2) it is surprising that catches of seahorses persist (Chapter 3), but because fishing activities are restricted to shallow areas (Chapter 4), deeper waters may provide natural refugia for exploited species. Therefore, in Chapter 5, I examine whether seahorses have a depth refuge from lantern fishing and assess the

importance of the refuge for maintaining population size and catch of the targeted species, *H. comes*. I construct a deterministic population accounting model that integrates available life history information with estimates of current exploitation rate (Chapter 4) to predict population sizes, extinction risk, and fishery benefits under different assumptions about the size of the refuge, amount of dispersal between fished and refuge zones, density dependence, and natural mortality.

Finally, I end with a general discussion of the findings presented in this thesis, and how those findings contribute to reconciling fisheries with conservation for small-scale, data-poor fisheries. I also draw upon my new understanding of the sustainability of the lantern fishery to make recommendations for future research, management, and conservation priority actions.

**Chapter 2: How does the accuracy of fisher knowledge affect  
seahorse conservation status?**



## **Introduction**

Historic baselines for the world's ocean fisheries are largely unknown (Pauly, 1995) and it is generally impractical to gather basic information on the many unstudied regions (Johannes, 1998). Formal scientific data required for managing many marine resources have not been gathered. Moreover, the large scale of most exploited areas means that controlled and replicated experiments are impossible (Ludwig *et al.*, 1993). Catch rates (catch per unit of effort – CPUE) have been used in formal fishery assessments as proxies of changing fish abundance, but have only been systematically collected for commercially important industrial fisheries (FAO, 2009). Small-scale fisheries that employ 98% of the world's fishers (Berkes *et al.*, 2001) and land more than half the world's annual marine catch (Davy, 2000) are poorly understood because there is little formal research or monitoring of exploitation.

Careful accumulation of fishers' knowledge is beginning to fill gaps in our quantitative understanding of the threatened status of marine species and ecosystems. We know from government reports, local knowledge and fisher logs that the Chinese bahaba (*Bahaba taipingensis*) and giant humphead parrot fish (*Bolbometopon muricatum*) have been fished to the brink of extinction (Sadovy & Cheung, 2003; Dulvy & Polunin, 2004) and current cod stocks (*Gadus morhua*) on Canada's Scotian Shelf are a small fraction of their historic numbers (Rosenberg *et al.*, 2005). In Indonesia, analysis of local knowledge suggests ecosystem-wide decline of exploited coral reef species (Ainsworth *et al.*, 2008).

Despite a growing interest in incorporating fisher knowledge into quantitative fisheries and conservation assessments, there remain practical impediments to its use. When

compared directly, fisher and scientific knowledge have been found to both agree (e.g. Neis *et al.*, 1999b; Begossi, 2008; Lozano-Montes, Pitcher & Haggan, 2008) and disagree (e.g. Otero *et al.*, 2005; Daw, 2008a), raising concerns about the accuracy of fisher knowledge. Work reviewed by Daw (2008a) illustrates that it is easier to recall events that are unusual or rare, especially if they were positive (e.g. a particularly lucrative fishing excursion), than other less striking circumstances (Bradburn, Rips & Shevell, 1987; Tourangeau, 2000). Rare, positive events may thus be overemphasized in retrospective interview datasets. ‘Retrospective bias’ has been recognized in recording historic fisher perceptions (Neis *et al.*, 1999b; Ainsworth *et al.*, 2008), but to our knowledge there are no reports on how different assumptions of bias affect quantitative assessments.

We report our efforts to address retrospective bias in CPUE data for a small-scale, data poor seahorse fishery in the Philippines. Our work had three objectives: (1) reconstruct CPUE using fisher interviews and daily logbooks; (2) consider how assumptions about the accuracy of fisher recall affect the assessment of conservation status for that species; and (3) compare current and past conservation status. We do not test the reliability of interviews by directly comparing them with logbooks because the datasets did not overlap in time; instead, we explore simple calculations of CPUE for logbooks that might account for recall bias in interviews, allowing us to link interview and logbook data into one time series.

Seahorses are collected as a cash commodity for export as non-food items, either dried for use in traditional medicines and as curiosities, or live for the aquarium trade. They have been traded internationally for decades, but we lack understanding of the status of most populations, hence we turn to fisher knowledge. Almost all seahorses are caught in small-scale fisheries that are not formally monitored or managed, but trade analyses led to

estimates of at least 20 million dried animals per year in 1995 (Vincent, 1996). Furthermore, all qualitative indicators of overfishing reveal that seahorses are overfished (Vincent, 1996; Martin-Smith *et al.*, 2004). The IUCN Red List (IUCN, 2008a) classifies extinction risk for 33 seahorse species: seven as Vulnerable (including our study species *Hippocampus comes*), one as Endangered, and 25 as Data Deficient (i.e. having insufficient data to be classified).

The Philippines has many unmanaged and unrecorded fisheries and has historically been a dominant source of seahorses in international trade. Almost all of the millions of seahorses exported (Vincent, 1996) are obtained in the small-scale fisheries that engage 90% of Filipino fishers (DA-BFAR, 2004). Exports have been recently curtailed by domestic legislation that has inadvertently (as a consequence of international trade restrictions) prohibited all seahorse exploitation since 2004. Nonetheless, domestically illegal fishing continues because there is little enforcement, seahorses command a high price on the internationally legal market, and fishers have few alternative sources of income. Quantifying the extent of declining seahorse catch rates is one way to estimate key parameters needed to evaluate effects of possible recovery options.

## **Methods**

### **Study area, fishery, and target species**

Our study takes place in the Danajon Bank region of the Central Philippines (Fig. 2.1). Destructive fishing has degraded this once-rich coral reef system to the point where marine life is grossly depleted. The people living in the region's coastal communities are

among the poorest in the Philippines. In 2002, 80% of fishing households in Northern Bohol fell below the poverty threshold (Green *et al.*, 2003). Our focal community, Handumon, has the largest concentration of lantern fishers who catch the most seahorses in the region (Meeuwig *et al.*, 2003). Project Seahorse has been working with the Handumon community since 1996 as part of a larger initiative to empower them to manage their marine resources sustainably.

Although the lantern fishery is the most extensively studied seahorse fishery, it is understood solely through fisher-derived knowledge and trade records. Lantern fishers swim at night under a kerosene lantern, and free-dive for seahorses, other finfish and invertebrates. This open access, small-scale, multi-species fishery has not been formally monitored or managed, but fishers and traders have reported declines in seahorse availability (Vincent, 1996; Pajaro *et al.*, 1997), increasing demand, price, and effort, suggesting overexploitation (Martin-Smith *et al.*, 2004). Analysis of the recent fishery suggests current CPUE is seasonal and much lower than the anecdotal fisher reports of historic CPUE (Vincent *et al.*, 2007). The historic anecdotes and magnitude of the decline have not yet been thoroughly assessed.

The tiger-tail seahorse, *Hippocampus comes* Cantor 1850, is the most frequently caught seahorse species in the lantern fishery (Vincent *et al.*, 2007). *Hippocampus comes* is found at night on coral reefs, sponge, and seaweed habitats (Perante *et al.*, 2002). It is listed as Vulnerable on the IUCN Red List (IUCN, 2008a) and studies of its life history, abundance, and distribution suggest that it fares poorly under high levels of fishing pressure (Morgan & Lourie, 2006; Morgan & Vincent, 2007).

## Analysis

To reconstruct the history of *H. comes* exploitation by lantern fishers we searched Project Seahorse unpublished data and reports. The only record of the earliest years of the fishery was contained in retrospective fisher interviews, whereas catch rates for more recent years were recorded in daily fisher logbooks. Because these two datasets do not overlap in time we cannot be sure how comparable they are. Interview data may be biased toward maximum catch nights, so we tried a number of calculations that could correct for retrospective bias linking interviews to logbooks as a single trend.

Although CPUE data may not be a reliable measure of changes in exploited populations (Harley *et al.*, 2001) we used it because: (1) it is the only metric of historic change available; (2) previous work suggested that it may be a robust indicator of *H. comes* population size (Vincent *et al.*, 2007); and (3) CPUE is an accepted proxy for abundance for IUCN Red Listing (IUCN, 2008b).

### *Interview CPUE (1970s to 1994) data collection*

From March to July 1995 Project Seahorse conducted extensive semi-structured interviews to document fisher understanding of the biology, fishery, trade, conservation and management of local seahorses. The second author, a Filipina, conducted interviews with 21 of the approximately 35 seahorse fishers in Handumon in the local Cebuano or Filipino language. All fishers were invited to be interviewed; all who were willing and available were

included. Their individual fishing experience ranged from six months to 24 years, covering the range of experience of all fishers.

From these interviews, we extracted only respondent biographical information and answers to three structured questions focused on understanding fishers' perceptions of historic change in the seahorse fishery between 1969 and 1993 (Table 2.1). Fishers were asked if CPUE had increased or decreased in recent years (Q1) and completed trend diagrams (Deguit *et al.*, 2004) representing relative change in CPUE over their career (Q2). Fishers were also asked to recall a typical catch night<sup>-1</sup> for a few specific years (Q3). Years were chosen to correspond with events that might aid fishers' recall, e.g. change of presidents, large typhoons, or something personal like the birth of a child (Means & Loftus, 1991). The number of responses to Q3 varied by time period (mean  $\pm$  1 S.D.: 12  $\pm$  7), with fewer fishers still alive and able to recall earliest catches than for more recent years. Our best estimate of the total number of seahorse fishers in the first decade of the fishery (roughly the 1970s) was between 5 and 10 individuals, hence the two fishers interviewed comprise between 20 and 40 % of all those who were fishing.

### ***Interview CPUE analysis***

To convert each fisher's trend diagram (Q2) into a relative percent change in career CPUE we totaled the number of X's for the initial year of a career (CPUE<sub>initial</sub>) and the most recent year of career (CPUE<sub>final</sub>). We then calculated change in career CPUE as (CPUE<sub>final</sub> – CPUE<sub>initial</sub>)/CPUE<sub>initial</sub>. We analyzed typical catch night<sup>-1</sup> (Q3) raw interview transcripts

by standardizing responses, compressing years into bins, and then calculating two CPUE metrics corresponding to different assumptions about retrospective bias.

(i) *Interview median*: With this metric, we assumed that when fishers were asked about ‘typical’ CPUE, they reported a measure of centre, with some error. We choose a median, rather than mean, because distributions of counts (number of seahorses) tend to be skewed, making the median a more suitable measure of central tendency. Fishers differed in how they provided ‘typical’ CPUE: some gave a range while others a single value. To standardize responses, when fishers responded with a range, we took the median value; when they gave a single value, we used that. We then grouped years into bins that were larger for earlier time periods. Because our goal was to look at long-term trends, and we assumed fishers’ ability to recall precisely which year a CPUE occurred would be worse for dates farther back in time (Bradburn, 2000), we grouped responses into four categories: 1994, 1990, during the 1980s, during the 1970s. We then calculated the median of all fishers’ CPUEs in a given time period to produce a single CPUE value for each time period.

(ii) *Interview max*: With this metric, we assume that fishers consistently over-estimated CPUE. We chose only the highest CPUE for each time period as the single yearly maximum CPUE.

### ***Logbook CPUE (1996 to 2003) data collection and analysis***

From 1996 to 2003, fishers worked with Project Seahorse to record daily seahorses catch in logbooks. Logbooks were completed by 43 fishers between 1996 and 2003 with a mean participation rate in the logbook program of  $16 \pm 5$  (1 S.D.) fishers year<sup>-1</sup> (Vincent *et al.*, 2007). To minimize bias we present results only for years that had equal sampling during peak and lean seasons. We then did three calculations to explore how we might ‘correct for’ biases in interview data allowing us to link interview and logbook data into a single trend:

- (i) *Logbook median*: For each fisher, we calculated the median catch night<sup>-1</sup> for each year from logbook data. We then took the median of these catches per fisher to produce a single CPUE value for each year.
- (ii) *Logbook median no zeros*: We repeated the calculations used for Logbook median, but removed nights on which zero seahorses had been caught.
- (iii) *Logbook max*: We took the highest catch night<sup>-1</sup> for each year as the single yearly maximum.

### ***Evaluating extinction risk***

Drawing on CPUE trends calculated from interview and logbook data we evaluated the extinction threat for *H. comes* using the IUCN Red List ([www.redlist.org](http://www.redlist.org)) criteria (IUCN, 2001). We calculated the percent decline in interview and logbook CPUE from the start to



the end of three time periods, the most recent of which (1990 to 2001) spanned interview and logbook datasets. We compared results to IUCN criteria to assign extinction risk. We focused on criterion A which measures the ‘observed, estimated, inferred, or suspected’ decline in the number of mature individuals over a ten year period or three generations, whichever is longer. Information needed to assess exact generation time as prescribed by the IUCN (2008b) is unavailable for *H. comes*. Therefore, we choose the default ten year time period because it is longer than three times the best estimate of maximum lifespan (Morgan, 2007) for this species.

We were particularly concerned with criterion A2, in which the causes of decline may not have ceased, may not be understood or may not be reversible. In these contexts, a decline of >30% would see a species listed as Vulnerable, >50% as Endangered and >80% as Critically Endangered. Since we were unable to ascertain the number of ‘mature individuals’ over the entire history of the fishery, we used the tally of all fish. Limited to the years with data available for this calculation, our time periods were either slightly less or more than ten years and the final year of one time period became the starting point of the following time period.

## **Results**

### **Interview CPUE (1970s to 1994)**

Fishers consistently reported declines in CPUE. All fishers said their CPUE in the past two years and past five years had declined. All changes in career CPUE were also

declines, ranging from 25-80% (mean 67%) with the maximum decline noted by a fisher who had been fishing since the 1970s. When asked specifically to recall numbers, fishers reported collecting a median CPUE of 150 seahorses fisher<sup>-1</sup> night<sup>-1</sup> at the outset of the market for seahorses in 1969. This had declined to ten seahorses fisher<sup>-1</sup> night<sup>-1</sup> by 1994, a 93% reduction. Max CPUE followed a similar trend falling from 200 to 50 seahorses fisher<sup>-1</sup> night<sup>-1</sup> (75% decline) over the same time period (Fig. 2.2). One fisher mentioned that he was motivated to start catching seahorses because two other fishers could catch more than 100 seahorses night<sup>-1</sup> in the 1970s and early 1980s. Only five fishers reported having any zero seahorse catch nights.

### **Logbook CPUE (1996 to 2003)**

From fisher logbooks we recorded a total of 31,381 *H. comes* caught over 11,179 nights of fishing from 1996 to 2003 with catches ranging from 0 – 80 seahorses night<sup>-1</sup>. We saw no clear trend in CPUE over the course of the logbook program (Fig. 2.2), although there was variability among years. This was true of all metrics, with or without zero catches, although absolute values varied according to the metric. Logbook data showed that catch night<sup>-1</sup> was highly skewed, with no seahorses on nearly 70% of fishing nights (Fig. 2.3a). Medians were reduced as a result of the high number of low catch nights but there were also nights with large catches, in all years and for all fishers (Fig. 2.3b).

### **Entire fishery CPUE trend (1970s to 2003)**

Overall reconstructed CPUE showed a fishery that experienced a severe historic decline with more recently low and variable CPUE. Our assumption about fisher recall bias, and therefore our choice of CPUE calculation, affected the inferred magnitude of decline in catch rate. Even the most optimistic calculation, using maxima, yielded a 77% decline in CPUE from 1970 to 2003. When we assumed that interviews overestimated historic CPUE we chose logbook calculations based on maxima or medians that excluded zero catch nights (Fig. 2.2). This choice revealed a fishery that was in decline from 1970 to 1993, but has been stabilizing at lower levels since 1996. When we assumed that interviews accurately portrayed historic CPUE we chose the logbook median CPUE calculation that included zero catch nights. This choice of CPUE calculation suggested that the fishery had collapsed rather comprehensively.

### **Evaluating extinction risk**

Extinction risk classification depended on the assumed reliability of fisher recall and the time period of data used. Different assumptions about recall bias, calculated as a means of linking interview and logbook data, translated into a wide range of extinction risk classifications for the most recent decade (1990-2001: Table 2.2). For example, when we assumed that interviews overestimated historic CPUE and then calculated a metric that might make logbook comparisons valid (Logbook max) we concluded that CPUE was increasing

and therefore this species should be classified as Least Concern. On the other hand, if we assumed that interviews and logbooks were comparable without ‘correction’, and calculated a median to summarize their central tendencies (Interview median plus Logbook median), we concluded that CPUE had declined by 97% during the 1990s and that *H. comes* should be listed as Critically Endangered.

When we compared historic with current extinction risk for *H. comes*, we found that for earlier decades extinction risk was consistently more worrying. Declines between 50 and 80% justified the classification of Endangered for both 1970-1980 and 1980-1990 (Table 2.2), regardless of how accurate we assumed interviews to be. In contrast, since 2002 *H. comes* has been listed only as Vulnerable on the IUCN Red List (2008a).

## **Discussion**

Handumon lantern fisher interviews tell the story of a seahorse fishery that experienced severe historic decline. The severity of that decline (and therefore extinction risk) hinges on assumptions about the accuracy of fisher recall. Fisher interviews set an historic baseline which suggests that *H. comes*, and therefore other historically exploited species, may be more at risk than current Red Listing reflects.

Fishers perceived historic declines in CPUE that were not apparent in more recent logbook trends. Fisher interviews provided the only record of historic catches and suggested that CPUE had declined 75 to 93% by 1994. Without fisher interviews we might have concluded that recent low CPUE from logbooks was also typical in the past. It is clear from

the consistent declines reported by all fishers and across triangulated questions that all fishers perceived a decline. Declining catch rates revealed by interviews are similar for other marine resources in Danajon Bank (Green *et al.*, 2004) and other fishery declines in the Philippines (Stobutzki *et al.*, 2006). Fishers' concerns over dwindling seahorse CPUE catalyzed the establishment of a community-managed no fishing zone in the hopes of recovering seahorses and other target species (Pajaro *et al.*, 1997). Certainly any management of seahorses should consider declines perceived by fishers to avoid the shifting baseline syndrome (Pauly, 1995).

How perceived historic declines relate to recent logbook trends depended on assumptions we made about fisher recall bias. Nights on which fishers caught zero seahorses dominated logbook data (Fig. 3.3a), but were mostly absent from interviews, making us wonder why fishers did not report zero catch nights. It is possible that if seahorses were more abundant historically fishers caught them every night, but it is also likely that fisher reports were biased toward positive rare events (Bradburn *et al.*, 1987; Tourangeau, 2000). Combined with the possibility that fishers may err toward pessimism (Ainsworth & Pitcher, 2005), fisher interviews likely tend to exaggerate the perceived decline. Although there are no standard methods for quantifying retrospective bias, our calculations show that if we assumed fishers overestimated historic CPUE, the decline to more recent CPUE in logbooks was less severe (Interview Max with Logbook Max: Fig. 2.2). On the other hand, if we assumed fisher historic interviews were accurate, the decline to present was more dramatic (Interview median with Logbook median: Fig. 2.2).

In addition to recall bias, interviews and logbooks generated data with different characteristics, requiring careful consideration during analysis. Logbooks captured more variability than interviews, which influenced conservation assessments. Logbook data had

more samples per time than interview data: more nights in a year and more years in a row. It is then not surprising that logbooks captured a wider spectrum of events from common to rare, zero to many seahorses caught respectively. By extension, extinction risk calculated from logbooks (1990-2001) spanned a wider range than when calculated only from interview data (1970-1980 and 1980-1990 Table 2.2).

Despite uncertainties about how best to link interviews and logbooks, our analysis suggests that our focal population of *H. comes* may be more at risk of extinction than its current listing of Vulnerable. Our uncertainty about the compatibility of interviews and logbooks placed our Red List assessment for the period 1990 - 2001 between Least Concern and Critically Endangered (Table 2.2). Least Concern is an unrealistic classification because it is based on a single max CPUE value per year, probably a rare value and possibly an outlier. *Hippocampus comes* should therefore be classified at least as Vulnerable, but possibly as at risk as Critically Endangered (CR). In situations where the spread of plausible values qualifies a taxon for two or more categories of threat, the IUCN recommends adopting a precautionary, but realistic, approach and choosing more threatened category (IUCN, 2008b). In our study, the more threatened choice of CR is based on data that assumes historic CPUE from interviews was not over estimated relative to logbooks. Clarification of this classification requires interviews that span the entire history of the fishery. The current listing of Vulnerable (IUCN, 2008a) is supported by our calculations that account for retrospective bias (Median interview and Median no zeros: Table 2.2), but may underestimate actual threat level, especially if the biggest declines occurred prior to 1990.

For species like *H. comes*, that have undergone historic exploitation, our results suggest that relying on recent decades of data alone may underestimate extinction risk.

Ignoring historic data, the current IUCN Red List assessment is Vulnerable. In contrast, assessment of the earliest data qualifies this species as Endangered. While a similar pattern has been found for Gulf grouper (*Mycteroperca jordani*; Sáenz-Arroyo *et al.*, 2005), the opposite pattern has been found for amphibians (Stuart *et al.*, 2004). For marine species that have been impacted by fishing for at least hundreds of years (Jackson *et al.*, 2001), assessment of contemporary conservation and fishery status must be conducted relative to historic baselines (Pauly, 1995). Our analysis highlights a potential limitation to the Red Listing process. Because the default time-frame for assessment is only a 10 year window, there is little capacity for incorporating long-term changes for relatively short-lived species. Capturing historic changes is especially important for species that have experienced historic declines. This type of retrospective threat classification is uncommon, but is becoming possible by incorporating fisher knowledge.

We have focused on examining retrospective bias to improve methods for incorporating fisher knowledge into quantitative assessments, but other questions about the magnitude of decline have not yet been answered. It is perhaps surprising that despite severe perceived declines in CPUE, there are still seahorses to be caught. Certainly the flexible, multi-species nature of the fishery allows lantern fishers to continue taking seahorses whenever they see them (Vincent *et al.*, 2007) and establishment of no fishing zones may allow for un-fished individuals to seed fishing grounds. In addition to retrospective bias, another way by which declines may be over-estimated is that some deep-water dwelling seahorses are less vulnerable to fishing because lantern fishers only fish in relatively shallow water. If deeper seahorses were not being fished, fishers would perceive their reachable stocks declining at a greater rate than the overall population. This bias applies to

scientifically gathered CPUE data as well and is known as hyperdepletion. We cannot yet assess this problem due to lack of data.

Our study that links two historic fisher-derived datasets can inform management, prioritize future research, and provide insight into how we might tackle challenges of integrating fisher knowledge into quantitative assessments. Fishers perceive severe historic declines in seahorse catch rates, which means seahorses may be more endangered than we think because current Red listing doesn't incorporate historic status. The success of future management and avoidance of the shifting baseline syndrome requires better understanding of fisher perceptions of the decline. Before our CPUE reconstruction can be used to evaluate effects of possible management options we will need to clarify the magnitude of interview bias (e.g. trying to confirm the magnitude of CPUE in the early 1970s that only two fishers could recall) and improve our understanding of how CPUE relates to *H. comes* abundance. Future studies that incorporate fisher knowledge into quantitative assessments require: 1) clearly stated assumptions about fisher knowledge bias; 2) clear criteria to compare fisher knowledge collected with different methods, and; 3) evaluation of the impact of assumptions on assessments.



## **Tables and figures**

**Table 2.1. Interview schedule**

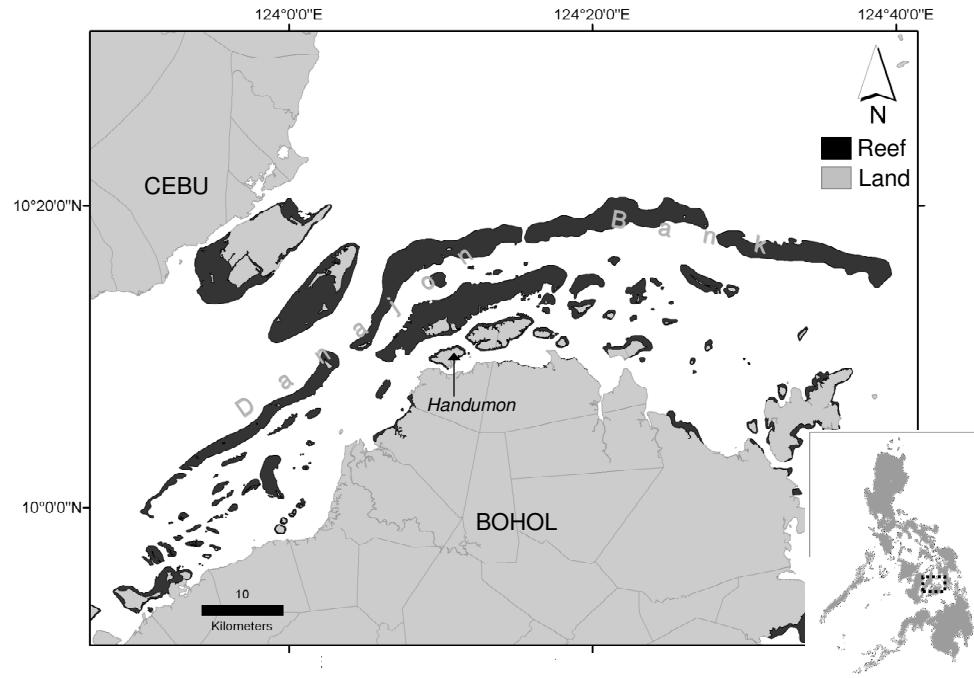
Biographical information: name, age, number years fishing, number years fishing seahorses
Q1. Has the number of seahorses you catch per night increased or decreased over the past two years? Over the past five years?
Q2. On this timeline (where we have established the beginning, middle, and most recent year of your seahorse fishing career) please mark a number of X's at each time period to represent relative change in CPUE.
Q3. What was the usual number of seahorses you caught per night in: 1994, 1990, 1980s, 1970s?

**Table 2.2. Extinction risk categorization for IUCN Red List criteria by time period and CPUE calculation: CR = Critically Endangered, EN = Endangered, VU = Vulnerable, LC = Least Concern**

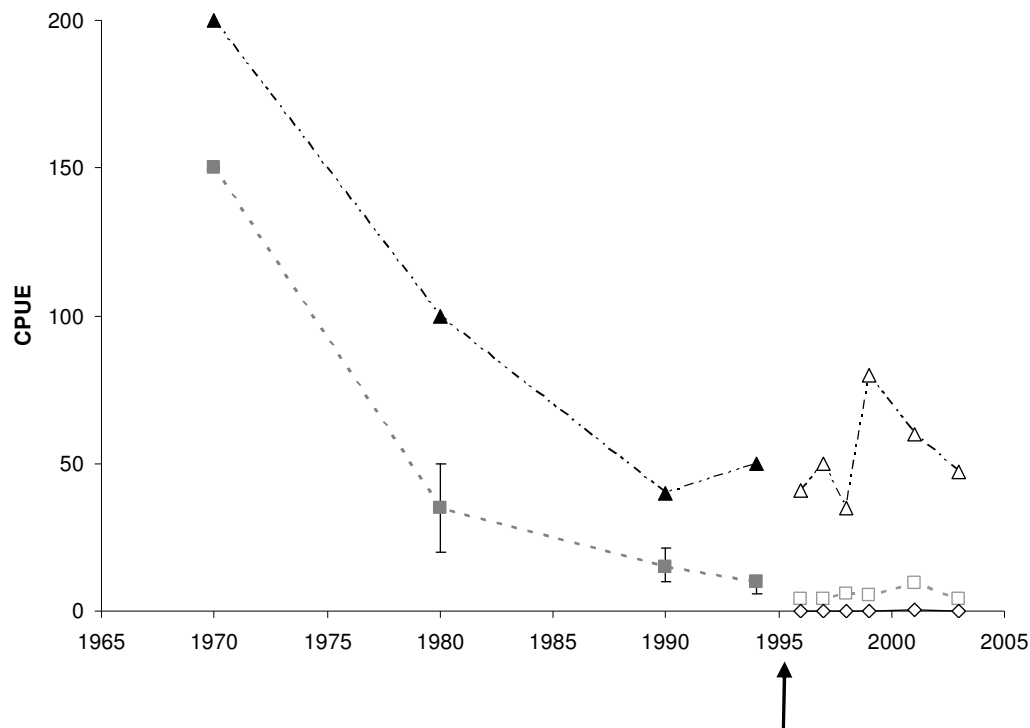
<i><b>CPUE Calculation</b></i>		<i><b>Time Period</b></i>		
<b>Interview</b>	<b>Logbook</b>	<b>1970-1980</b>	<b>1980-1990</b>	<b>1990-2001</b>
Max		EN	EN	
Median		EN	EN	
Max	Max			LC
Median	Median no zeros			VU*
Median	Median with zeros			CR

\*VU indicates current Red List classification

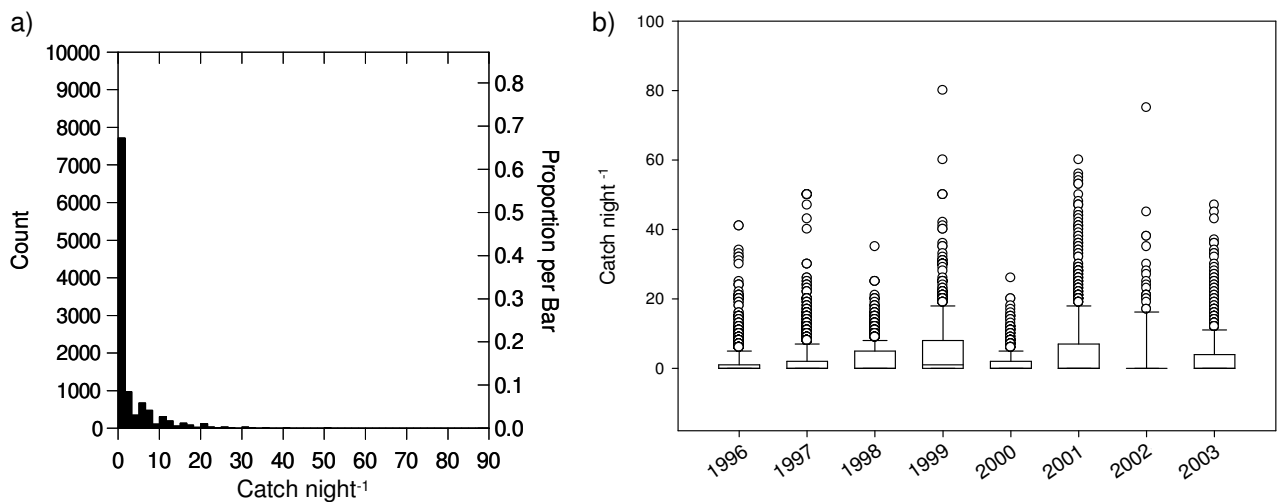
**Figure 2.1. Danajon Bank study region of the central Philippines and our focal lantern fishing community of Handumon**



**Figure 2.2.** All CPUE trend calculations for interview data (maximum: ▲, median: ■) and logbook data (maximum: △, median no zeros: □, median: ◇). Arrow indicates break between interview (1970 – 1994) and log book (1996 – 2003) datasets. All calculations are in units of seahorses fisher<sup>-1</sup> night<sup>-1</sup>. Error bars are 25<sup>th</sup> and 75<sup>th</sup> percentiles for median values calculated for all medians, but large enough only to be visible on the graph for interview medians.



**Figure 2.3.** Frequency (a) and distribution (b) of catch night<sup>-1</sup> logbook data. Box plot shows 25<sup>th</sup>, 75<sup>th</sup> percentile, and median; whiskers are 90<sup>th</sup> percentiles; hollow circles are outliers.



## **Chapter 3: Comparing fisher interviews, logbooks, and catch landings estimates of extraction rates in a small-scale fishery**

## **Introduction**

Rapid, low-cost, and reliable assessment methods are needed to detect trends in the abundance and use of wildlife. Data needed for managing wildlife exploitation (e.g. population trends, hunting patterns) have traditionally been gathered using formal scientific methods (Mauro & Hardison, 2000). However in developing countries with high biodiversity, heavy reliance on wildlife resources, and rapid environmental change, these time-consuming and expensive methods (Anadón *et al.*, 2009; Rist *et al.*, 2010) are often impractical (Johannes, 1998; Danielsen *et al.*, 2005). With the number of threatened species on the rise and the vast majority (>95%) of described species yet to be assessed (Vié *et al.*, 2009), traditional scientific methods alone are unlikely to be able to inform species assessments – a critical first step toward halting declines. This issue is particularly concerning in marine systems, where our understanding of historic conditions is poor (Jackson *et al.*, 2001).

To understand changes in wildlife abundance and exploitation, researchers have begun to turn to alternative sources of data. Trade records (Baum & Vincent, 2005; Scales, Balmford & Manica, 2007), historic photographs (McClenachan, 2009), explorers' records (Sáenz-Arroyo *et al.*, 2006; McClenachan & Cooper, 2008), and unpublished government reports or grey literature (Sadovy & Cheung, 2003; Dulvy & Polunin, 2004) have all shed light on the extent of depletion in marine systems. Documenting the knowledge of resource users (variously known as local ecological knowledge (LEK), traditional knowledge, etc.), in particular, can be an inexpensive (Moore *et al.*, 2010; Rist *et al.*, 2010) and effective way of gathering data on species, catches, ecology, and habitat (Johannes, 1981; Neis *et al.*, 1999b).

Interviews with resource users (e.g. Sáenz-Arroyo *et al.*, 2005) and analyses of their daily written logs (Baum *et al.*, 2003; Rosenberg *et al.*, 2005) have uncovered population declines that were previously undetected by western science (e.g. Gilchrist *et al.*, 2005).

Despite its practicality, there are several reasons why resource user knowledge is not more widely incorporated into conservation and resource management. Although the use of LEK is increasing, only a small fraction (0.42%) of papers in 12 prominent ecological and conservation journals incorporated LEK between 1980 and 2004 (Brook & McLachlan, 2008). Perhaps the most significant impediment is the difficulty in appraising the reliability of LEK in terms of accuracy and precision. Information collected from resource users is subject to various and often unknown biases (reviewed by Daw, 2008a). Perceptions can be distorted through complex cognitive processes or vested interests (Bradburn *et al.*, 1987) and little is known about the quality of the information reported regarding events in the distant past. The reliability of interview data as indicators of trends is therefore unclear (Jones *et al.*, 2008). LEK assessments are also inter-disciplinary in nature and therefore either require appropriate training or collaborative work (Huntington, 2000). Furthermore, rigorous testing of LEK is necessary prior to its incorporation into management plans (Gilchrist *et al.*, 2005; Anadón *et al.*, 2009).

Studies that have assessed the consistency of information from resource users (e.g. interview or logbook data) and scientific approaches (e.g. landings records) have produced variable findings. In the marine realm, fisher testimony of trends in the abundance of corvinas (Sciaenidae: *Cynoscion* spp.) in the upper Gulf of California was in agreement with 50 years of recorded catch data (Lozano-Montes *et al.*, 2008). In contrast, northern British Columbia fishers reported steeper declines in multi-species abundance than were found in

stock assessments (Ainsworth & Pitcher, 2005). Compared to recorded landings, fishers may over- (Lunn & Dearden, 2006), under- (Rocha *et al.*, 2004) or accurately estimate (Neis *et al.*, 1999b) present catch or catch per unit of effort (CPUE) during interviews. Data in user logbooks led to inferences of lower catches (Walsh, Kleiber & McCracken, 2002) and higher effort (Rist *et al.*, 2010) than did scientific data collection. There is, therefore, a continuing need to understand the relationships between user-generated knowledge (Gavin & Anderson, 2005; Gilchrist *et al.*, 2005; White *et al.*, 2005) and other datasets (Gagnon & Berteaux, 2009).

Our objective was to evaluate the degree of consensus among three methods (fisher interviews, fisher logbooks, and “scientific” catch landings) of quantifying catch rates (catch per unit of effort or CPUE) from the same years for a small-scale, data-poor seahorse fishery in the Philippines. While other studies have evaluated a single resource-user based data collection method, we compare two data-collection methods that are based on concurrent resource user knowledge (interviews and logbooks) with scientific methods (catch landings) which is uncommon in the marine realm. Any differences or similarities among these modes of data collection will be important to consider when incorporating fisher knowledge into quantitative assessments (Neis *et al.*, 1999b; Huntington *et al.*, 2004; Gilchrist *et al.*, 2005) and when selecting practical data collection methods for conservation and fishery assessment.

Seahorses are globally exploited, but their fisheries remain data-poor. Almost 80 countries trade seahorses for use in traditional medicines, ornamental display, or curios (Vincent, 1996) yet we lack data to assess quantitatively the conservation status of 28 of the 36 seahorse species (IUCN, 2008a). Furthermore, traditional underwater survey methods are

time consuming and expensive for these small, cryptic, sparsely distributed fish (Foster & Vincent, 2004). There is, therefore, a pressing need for inexpensive rapid assessment methods that can detect trends in seahorse population size and exploitation rates to facilitate the management of these species.

## **Methods**

### **Study area, fishery, and species**

In the Philippines, an historically important exporter of seahorses (Vincent, 1996), the authors and their Project Seahorse team have worked with fishing communities since 1995 to assess and manage seahorse fishing. Our work is focused on the Danajon Bank in the Central Philippines (Fig. 3.1) - a double-reef system that has been largely degraded by destructive fishing practices. Coastal communities in the region are among the poorest in the Philippines (Green *et al.*, 2003). One of our focal communities, Handumon (municipality of Getafe, Bohol province), has the largest concentration of lantern fishers who catch the most seahorses in the region (Meeuwig *et al.*, 2003). Another focal community, the neighboring village of Jandayan Norte, has only a handful of elder lantern fishers who fish the same waters as Handumon fishers.

Seahorses are caught with many other species in a small-scale fishery that involves fishers free diving at night by the light of a lantern slung low over the bow of their outrigger boat. This open-access fishery has not been formally monitored or managed but fishers and traders have reported declines in seahorse CPUE (Vincent, 1996; Pajaro *et al.*, 1997);



coupled with reports of increasing demand, price, and effort, all available evidence suggests overexploitation (Martin-Smith *et al.*, 2004). Recent analysis of the fishery suggests current CPUE is extremely low and seasonal (Vincent *et al.*, 2007). Furthermore, fishers perceive historic declines in fishing rate that cannot be detected in recent (1996 to 2004) data, although the extent of the historic decline (and therefore extinction risk) hinges on the assumptions we make about the accuracy of fisher memory (O'Donnell *et al.*, 2010).

The seahorse targeted by lantern fishers is the tiger-tail seahorse *Hippocampus comes* Cantor 1850. This species comprises the majority of seahorses caught in the lantern fishery and in the Danajon Bank region overall (Vincent *et al.*, 2007). Although fishing seahorses has been illegal in the Philippines since 2004 - as a national reaction to international trade controls - fishers and buyers continue to trade openly. Project Seahorse has continued to obtain access to seahorse fishers by working through long-serving Filipino community organizers (who have engendered trust among villagers) and by maintaining open dialogue with fishing communities, emphasizing an interest in resource use rather than law enforcement. We also collected and retained our data in a way that coded for individuals, allowing them anonymity. As a result, most fishers and buyers spoke openly about seahorse catch and some volunteered to record catch in logbooks or have landings recorded. This allowed us to obtain and compare interview, logbook and landings data over the same time frame. We anchored our comparison in landings data, as representative of the scientific approach – whereas interview and logbook approaches both derive from local understanding – but this should not be taken to imply a greater confidence in that dataset.

## **Data collection**

### ***Interviews***

We conducted structured interviews with fishers from March to June 2008 in Handumon and Jandayan Norte (Fig. 3.1). Since there is no official register of fishers, we consulted village leaders for initial recommendations of individuals who might be willing to be interviewed and thereafter found respondents by word-of-mouth (Snowball sampling; Berg, 2001). Our goal was to gather information from all decades of the fishery. We achieved this by interviewing lantern fishers from neighboring communities that tend to fish the same grounds whose ages (24 – 66) and seahorse fishing experience (4 – 36 years), covered the range of all lantern fishers. Ultimately we interviewed 24 fishers from Handumon and 1 from Jandayan Norte (~50 and 30 % of the 2008 fleets, respectively).

Each interview began by constructing a detailed visual timeline that mapped out the fisher's personal history (Appendix A). This career history method (after Neis *et al.*, 1999b) guided interviews and increased the accuracy of recalled dates (Means & Loftus, 1991). The ensuing visualization allowed us to link fishers' important life events (e.g. first year fishing, marriage, birth of a child) to external events of known date (e.g. change of presidents, large typhoons). Subsequent questions asking fishers to recall CPUE were dated relative to the personal events mapped on the visualisation.

We asked fishers to recall a range of CPUE values during time periods when they were most likely to have clear memories. Memories of rare, especially positive, events and those that are more recent tend to be more readily recalled (Bradburn *et al.*, 1987; Tourangeau, 2000). We therefore asked individual lantern fishers about their catch on a

good, bad, and typical night of fishing during three time periods over their career: 1) the highest catch year, 2) the most recent year fishing, and 3) either a year early in their career or another year for which they had a clear memory of catch. Interviews were conducted in the local language using trained research assistants, and followed the interview schedule (Table 3.1). Fishers rarely reported zero seahorses caught per night, leaving ambiguity about whether they sought seahorses without success or never sought them at all.

### ***Logbooks***

From 1996 to 2003, Handumon fishers volunteered to record catch in logbooks. Logbooks were distributed by Filipino Project Seahorse team members who followed up with discussions and log collection at the end of each month. Of all the data recorded (refer to Vincent *et al.*, 2007 for full details) we focused on fishers' record of their nightly catch of seahorses. Logbooks were completed by 43 fishers between 1996 and 2003 with a mean participation rate in the logbook program of 16 ( $\pm 5$  S.D.) fishers year<sup>-1</sup>. This level of participation represented about 50% of the fishers in Handumon in a given year, with participation varying by month. Fishers seldom recorded whether they were trying to catch seahorses on a particular fishing expedition. Thus, nights or months with zero seahorse catch may have arisen either from an inability to find seahorses or from a decision to target other species. As logbooks were dominated by zero catch nights (Fig. 3.2) we also analysed a variant where we removed zeroes to make them more comparable to interviews and catch landings.

### ***Catch landings***

From 1996 to 1998 and 2002 to 2007 Project Seahorse collected biological data on seahorses caught by 42 lantern fishers in Handumon and five in Jandayan Norte with a mean participation rate of 29 ( $\pm 5$  S.D.) and 3 ( $\pm 1$  S.D.) fishers year<sup>-1</sup>, respectively. This level of participation represented about 80% of the fishers in Handumon and 60% in Jandayan Norte in a given year, with participation varying by month. Fishers are usually widely dispersed across their village and fishing grounds but predictably visit few buyers to sell their catch. Thus, to maximize sample sizes, data were collected at buyers' houses by a trained local assistant. For each seahorse they recorded the fisher and the species, size, sex, reproductive status (*sensu* Vincent *et al.*, 2007), number and location of capture. Since fishers only visited buyers when they had caught seahorses, we were unable to collect information on (or distinguish between) days when fishers did not fish or when they fished but caught no seahorses.

### **Analysis**

We compared and contrasted the three sources of CPUE (seahorses<sup>-1</sup>fisher<sup>-1</sup>night<sup>-1</sup>) trends and overall absolute values: interviews, logbooks and landings. Because of the positively skewed nature of the datasets and presence of outliers (Fig. 3.2), we used the median rather than the mean CPUE. All analyses were conducted using R (R Core Development Team, 2009).

## ***Trends in CPUE***

We evaluated the change in CPUE from 1996 to 2004 using linear mixed-effects models separately for four datasets: 1) interviews, 2) logbooks, 3) logbooks without zeros, and 4) landings. To compare like with like, we contrasted typical CPUE reported in interviews with median CPUE from catch landings, logbooks and logbooks excluding zeroes specific to each year. Only 8 of the 25 fishers interviewed (representing ~20% of the 1996 to 2004 fishing fleet) provided data that fell between 1996 and 2004 – the range of years for which logbook and landings data were available; the ages (26 - 66) and years fishing experience (10 – 36) of these 8 fishers were comparable to our larger pool of respondents. To obtain fisher-specific medians from logbooks, logbooks without zeroes, and catch landings, we calculated median CPUE for each fisher within each year (henceforth, ‘median CPUE’). For logbooks and landings, the number of nights fished by a fisher in a given year varied widely among fishers (from 1 to 250). Therefore, we also weighted each fisher’s median CPUE year<sup>-1</sup> by the number of nights fished in that year such that median CPUE values based on more nights fished were given more weight.

We used top-down model minimization (Diggle *et al.*, 2002) to choose the minimum-adequate model for each dataset. We could not develop one all-encompassing model with *method of data collection* as a random component since the datasets differed in structure. We began with a linear mixed-effects model with CPUE as our response variable, year as a fixed effect and fisher as a random effect on slope and intercept (executed in R with the nlme package; Pinheiro *et al.*, 2009). Random effects of fisher accounted for non-independence of CPUE data across years, and accounted for differences among fishers in their overall CPUE

(intercept) and in their change in catch through time (slope). Since a change can only be estimated with a minimum of two reports of CPUE, we excluded fishers for whom we had only a single year of data in logbooks and/or landings records. We then dropped random effects based on the results of likelihood ratio test (REML) that compared the fit of models with and without each term. Prior to analyses, all data were  $\ln(x+1)$ -transformed to fulfill model assumptions, which were verified visually following Zuur *et al.* (2009). The optimal model varied by data-collection method (Table 3.2).

### ***Absolute value of CPUE***

Since slopes of trend in CPUE were not different among data sources (see results), we compared overall CPUE estimated using each data-collection method, including fisher interview estimates of good and bad CPUE in addition to the median CPUE from previous analyses. Here we compared CPUE between 1996 and 2004 for six datasets: 1) interview – good nights, 2) interview – bad nights, 3) interview – typical nights, 4) logbooks - median, 5) logbooks without zeros - median, and 6) landings – median.

To compare the central tendencies of the data collection methods directly we tested the hypothesis that CPUE differed significantly among the six data-collection methods. Because no trend across years was detected (see results for median CPUE; interview good nights  $p = 0.31$ ; interview bad nights  $p = 0.39$ ) we pooled CPUE values for all fishers and all years within a data-collection method. We then conducted a repeated measures ANOVA (RM-ANOVA, Crawley, 2007) assuming a quasi-Poisson distribution. This test was

executed as a generalized linear mixed model with the lme4 package in R (Bates & Maechler, 2009). The response variable was CPUE, the categorical fixed explanatory variable was data collection method, and fisher was a random factor nested within method. Pairwise comparisons were made between landings and each of the other data collection methods. Since we were unable to find a follow-up multiple comparison test, we evaluated differences among model estimated means by using the  $\pm 2 \times \text{SE}$  criteria described by Payton *et al.* (2003). We considered non-overlapping confidence intervals to be significantly different.

### ***Correlation – interview and catch landings***

We plotted interview responses against recorded landings to understand the relationship between what a fisher reported catching and what someone else recorded him catching. Data consisted of a point for each of 14 fishers who both responded to an interview and had his catch landings recorded in the same year, for any year between 1996 and 2007 (longer than in other analyses). For interviews we used responses to a good, bad, and typical CPUE which we matched with the maximum, minimum, and median landings value for a particular fisher and year. Since data were not normally distributed, we used a Spearman's rank correlation to test if the ranks of interviews co-varied with the ranks of landings.

## **Results**

### **Data structure**

Median CPUE data provided from interviews, catch landings, and logbooks differed in terms of distribution (Fig. 3.2). While zeros were absent from interviews and catch landings, logbooks were dominated (66%) by nights with no catch. When zeros were removed from logbooks, data remained positively-skewed. The same was also true of landings data. Interview data were almost uniformly distributed between CPUE of 5 and 50.

### **Trends in CPUE**

All metrics indicated no change in median CPUE with no slope parameters significantly different from zero ( $p > 0.05$ , Table 3.2, Fig. 3.3). Quantile-quantile and plots of the standardized residuals vs. fitted values indicated that residuals and random effects were not normally distributed for logbook and logbook without zeros datasets, likely because model predicted values tended to over-estimate low observed values and under-estimate high observed values. Model residuals and random effects for catch landings were normally distributed, as were model residuals for interviews. We found no heteroscedasticity in residuals across years suggesting that models are equally good (or bad) at predicting CPUE across years.



### **Absolute value of CPUE**

Interviews tended to give higher and more variable CPUE values than the other concurrent data-collection methods (Fig. 3.4a), but fisher reports of a bad night's catch approximated median CPUE from other data collection methods. Fisher reports of a good night's catch were significantly higher than other data collection methods (Table 3.3). Typical CPUE from interviews was on average ~ 16 times greater than median CPUE from logbooks and double the estimate from landings. Typical CPUE from interviews did not differ significantly from logbooks without zeros (Fig. 3.4b, Table 3.3). Interviews aside, CPUE from landings was only comparable to logbook estimates when zero-catch nights were not considered. Median CPUE recorded in logbooks was significantly lower than recorded in landings (Table 3.3), but when we removed the zeros from logbook data this difference disappeared. The assumption of homogeneity of variance was violated for this test because fishers provided CPUE values that varied more widely when responding to interviews (Levene's test:  $F=39$ ,  $d.f.=3$ ,  $p<0.001$ ) than were recorded in logbooks or landings.

### **Correlation – interview and catch landings**

There was no consistent relationship between interview reports and landings. The tightest relationship, though not significant, was between what fishers reported as a good night's catch and their maximum landings CPUE ( $r = -0.48$ ,  $p = 0.09$ ,  $n = 14$  fishers). There was no correlation between fisher reports of bad night's catch and recorded minimum CPUE

( $r = -0.074$ ,  $p = 0.80$ ,  $n = 14$  fishers) or a typical night's catch and recorded median CPUE ( $r = -0.048$ ,  $p = 0.87$ ,  $n = 14$  fishers). Generally minimum and median landings CPUE were roughly 5 seahorses night<sup>-1</sup>. In comparison, fisher reports of bad CPUE in the same year ranged from zero to 20 seahorses night<sup>-1</sup> while typical CPUE ranged from 11 to over 40 seahorses night<sup>-1</sup>.

## **Discussion**

Whether lantern fisher knowledge was comparable to other available CPUE data depended on which method (interview or logbook) was used to elicit knowledge and the variable (trends or absolute values) measured. While interviews and logbooks provided estimates of CPUE trends consistent with landings, interviews yielded higher and more variable estimates that were not correlated with landings. In spite of these discrepancies the lantern fishery, like other data-poor and depleted fisheries, is in need of recommendations for how to manage exploitation. We suggest two possible criteria to choose which methods might produce data appropriate for setting recovery or fishery targets.

No data collection method identified a change in CPUE between 1996 and 2004, a finding that is supported by fishery independent data. Despite biases associated with recalling past events (reviewed by Daw, 2008a) and the illegality of the fishery, fisher interviews produced results not so different from other methods. Underwater visual census (UVC) surveys of the tiger tail seahorse conducted from 2001 to 2008 also found stable densities both inside Marine Protected Areas (MPAs) and in fishing grounds (Yasue *et al.* in

prep). These results suggest that interviews or logbooks may be a reasonable proxy for more costly landings or UVC surveys when seeking information about trends. We cannot rule out, however, that trends went undetected by all methods. Increasing the ‘grain’ of effort from catch per night to weeks or months could decrease variability in CPUE (Anadón *et al.*, 2009), perhaps increasing the power of all data-collection methods to detect trends.

The generally higher CPUE reported in interviews contradicted our expectation that these data would agree with logbook records, also collected by fishers. The discrepancy seemed to arise from the nights with zero seahorse catch, which were rare in interview recall but dominated logbooks; bad or typical CPUE reported in interviews produced the same results as data from logbooks without zeros. Fishers may more readily recall rare and/or positive events (e.g. a night when many seahorses were caught) than common occurrences (e.g. nights of zero seahorse catch) (Bradburn *et al.*, 1987). Logbooks, in comparison, may be less biased if CPUE is recorded when it occurs, minimizing the reliance on memory. The fact that landings most closely approximated fisher reports of a bad nights’ catch supports an inference that interviews tend to over-estimate CPUE.

Interview reports of higher CPUE than other datasets identify may represent an over-reporting by the fisher or an under-recording in logbooks or landings data. There are three types of explanation for any over-reporting. First, seasonal variation in seahorse CPUE (Vincent *et al.*, 2007) may make it challenging for lantern fishers to estimate a typical value for catch night<sup>-1</sup> (Jones *et al.*, 2008). Second, fishers may report peak-season CPUE as typical, using imperfect recollection (Tourangeau, 2000). Third, fishers may intentionally over-report catches to impress an interviewer, to comply with perceived interview attitudes,

or perceived opportunities to attract outside investment to their communities (reviewed by Yasué *et al.*, 2010).

The other explanation for the discrepancy between interview and landings data may lie in a possible under-estimate of both effort and catch. First, by focusing our sampling at buying stations where fishers congregate – the dispersed fishing activities of small-scale fishers are difficult to track – our landings data excluded nights when no seahorses were caught. Second, lantern fishers sometimes sold fish at sea before returning to shore (as in Young *et al.*, 2006) gave fish as gifts before selling the bulk of the catch to the buyer or, on occasion, kept fish for personal use; in all cases, they would escape detection in landings data.

The discrepancy between what fishers reported (interviews) and what they were documented as catching (landing records) has important consequences for conservation and fisheries analysis. Although other studies have found strong correlations between interview and formally collected data (Anadón *et al.*, 2009; Rist *et al.*, 2010), the overall correlation depends on the consistency of individual relationships between interview recall and other datasets; if all individual relationships have similar slopes, then the overall correlation should be obvious whereas a myriad different slopes would obscure any correlation. The latter could arise if, for example, fishers differed in their abilities to recall high catches, low catches or both. Therefore, even if individual fisher's interview responses were correlated with official records, it may not be possible to quantify how fisher reports relate to scientifically collected data at a scale (the fishery) relevant to management/conservation.

Despite discrepancy among data-collection methods, managers who need to know current CPUE to set recovery / fishery targets could choose based either on corroboration

among methods or precaution. In the lantern fishery, a bad night's catch as reported in interviews, landings, and logbooks stripped of zeros all provided estimates of CPUE in the range of 1 to 10 seahorses fisher<sup>-1</sup> night<sup>-1</sup>. The disadvantage of basing a decision on consensus, in this case, is that none of these methods accounted for fishing effort accurately; nights when no seahorses were caught were not incorporated. They therefore overestimate CPUE, potentially leading to higher catch limits or less area placed in MPAs if management is deemed necessary. A more precautionary manager would base management on logbook data because as the only method to account for all fishing effort, including nights when no seahorses were caught, they produced the lowest CPUE values (just 0 to 1 seahorses fisher<sup>-1</sup> night<sup>-1</sup>). This choice would lead to lower catch limits or more area placed in MPAs than the consensus method.

For researchers seeking rapid assessment methods for trends in the abundance and use of wildlife, our study provides two suggestions. First, if interview data are used to set recovery targets or catch limits, which are sensitive to the magnitude of reported values, it will be important to ask about extremes (e.g. good and bad catches) in addition to typical or average values. Second, due to the unpredictable relationship between resource user knowledge and other datasets, collecting and then comparing multiple datasets will produce the most robust information. Even if data elicited from resource users can not directly inform quantitative assessments, they will continue to be important to understand resource user perceptions to effectively engage communities in management and conservation.

## Tables and Figures

**Table 3.1. Interview schedule.**

**Biographical information:** name, age, number of years seahorse fishing

**Personal timeline questions** (year): began fishing, married, children's births, purchased boats or engines

**CPUE questions:**

*During this past year...*

How many seahorses could you catch on a good night? A typical night? A bad night?

*When was the time you remember catching the most seahorses in one night?*

How many did you catch on a good night back then?

During that same time, what was a typical catch/night? A bad catch/night?

*At the beginning of your fishing career (or another year for which the respondent had a clear memory)...*

How many could you catch on a good night? A typical night? A bad night?

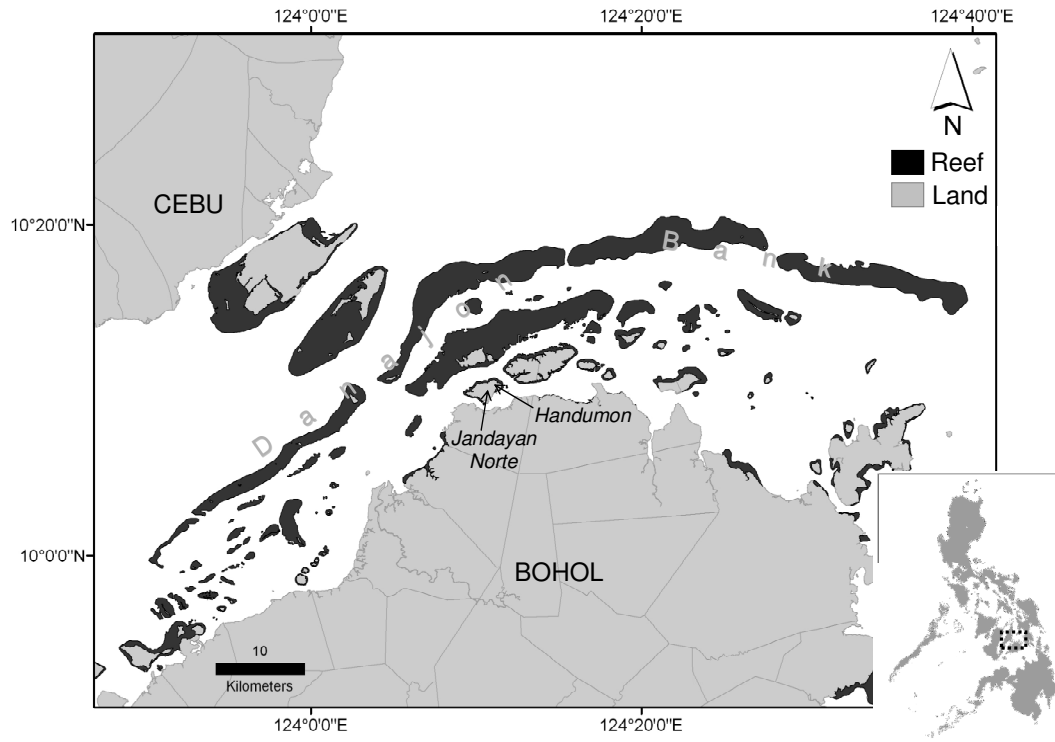
**Table 3.2. Sample sizes (n: total fishers) and best linear unbiased predictor model fit parameters to median CPUE date for 1996 to 2004.**

Method	Model	n	Slope		Intercept	
			value	p	value	p
Interview	Linear	8	0.062	0.64	2.78	0.002
Logbook	Linear mixed effects, random slope and intercept	43	0.007	0.91	0.61	0.004
Logbook without zeros	Linear mixed effects, random slope	39	-0.005	0.76	1.73	<0.001
Catch Landings	Linear mixed effects, random slope and intercept	42	-0.018	0.23	1.66	<0.001

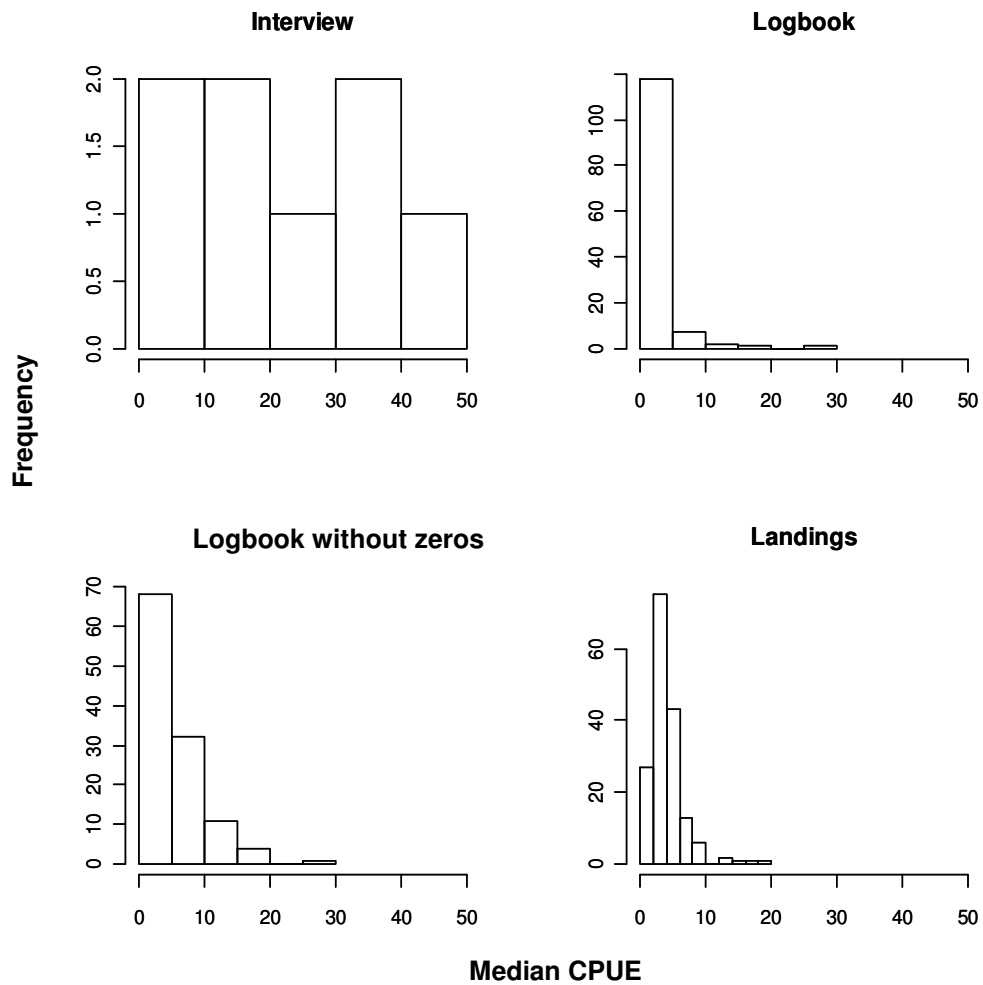
**Table 3.3. Sample mean ( $\bar{x}$ ) in seahorses fisher<sup>-1</sup> night<sup>-1</sup>, Ln + 1 and confidence interval (2 x SE) values for RM-ANOVA analysis by data-collection method.**

Method	$\bar{x}$	2 x SE
Interview good nights	3.983	1.089
Interview bad nights	1.937	1.228
Interview typical nights	3.018	1.122
Logbook without zeros	1.647	0.665
Catch Landings	1.411	0.415
Logbook	0.191	0.735

**Figure 3.1. Danajon Bank study region of the central Philippines and our focal lantern fishing communities of Handumon and Jandayan Norte.**

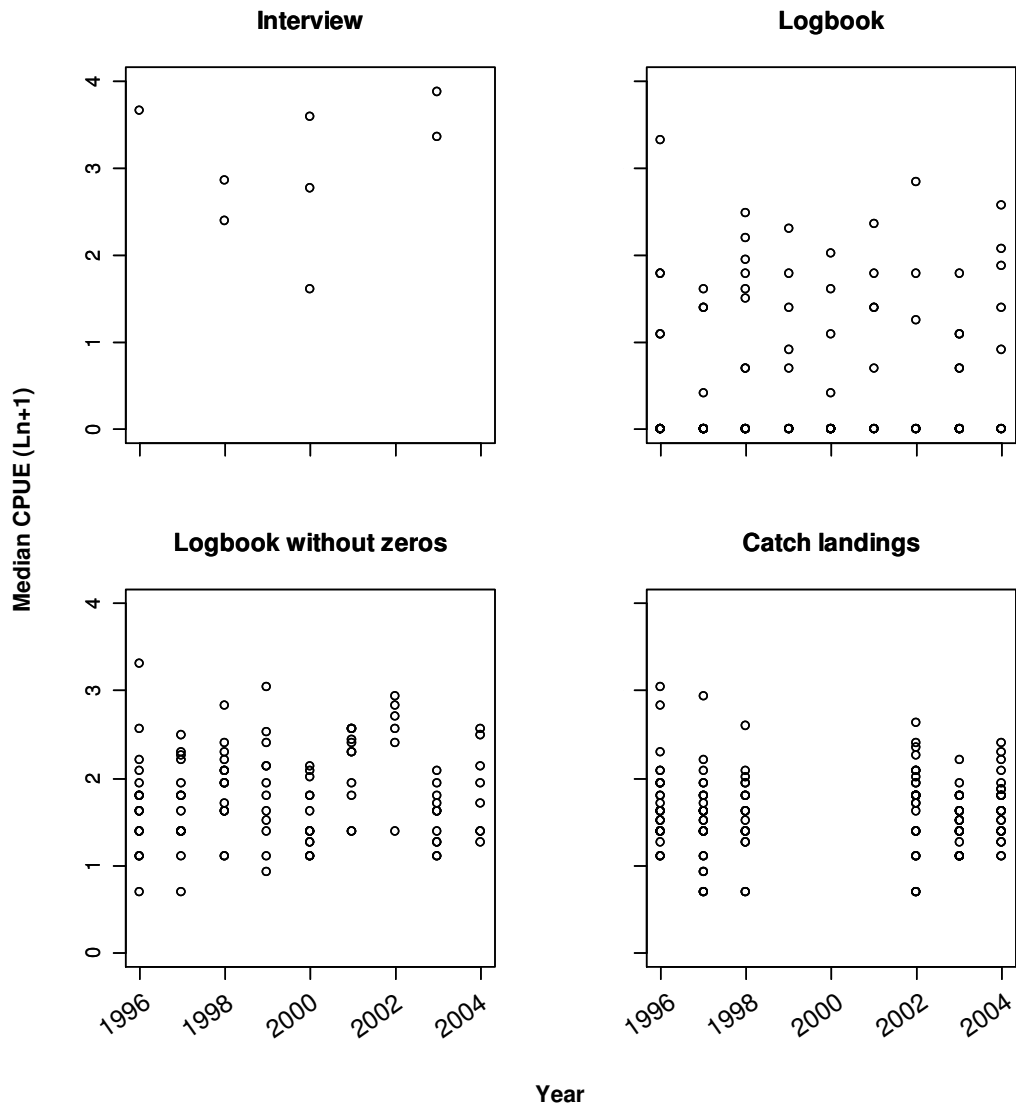


**Figure 3.2.** Frequency of median CPUE (seahorses fisher<sup>-1</sup> night<sup>-1</sup>) for interviews, logbooks, logbooks without zeros, and catch landings 1996 to 2004.

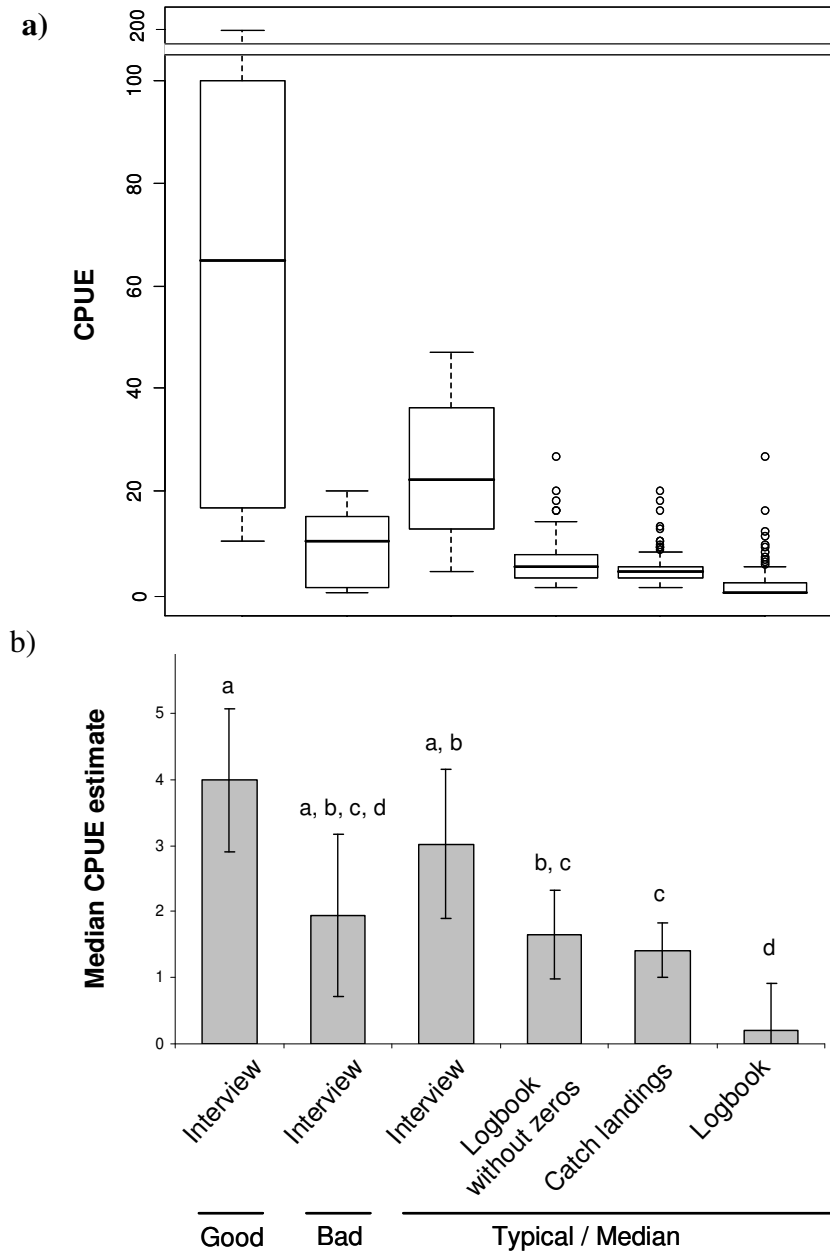




**Figure 3.3.** Median CPUE (seahorses fisher<sup>-1</sup> night<sup>-1</sup>, Ln + 1) used to assess trends from 1996 to 2004 for interview, catch landings, logbooks and logbooks without zeros data-collection methods. Each point represents an individual fisher.



**Figure 3.4.** Comparison of pooled 1996 to 2004 CPUE (seahorses fisher<sup>-1</sup> night<sup>-1</sup>) values (a) and RM-ANOVA estimate of CPUE (seahorses fisher<sup>-1</sup> night<sup>-1</sup>, Ln + 1) ± 2 SE (b) for fisher estimates of a good, bad, and normal night's catch from interviews and median CPUE from logbooks without zeros, catch landings, and logbooks.



**Chapter 4: High fishing mortality rate in a small-scale fishery: digital tracking of Filipino lantern fishing.**

## **Introduction**

While international attention to global fishery problems has focused on industrial fisheries (e.g. Pauly *et al.*, 2002; Myers & Worm, 2003; Castilla & Defeo, 2005), small-scale fisheries are increasingly suggested as a sustainable alternative, though they are poorly understood (Munro, 1996; Orensanz *et al.*, 2005). The distinction between these two types of fisheries tends to be by boat size: industrial referring to large, open-ocean vessels and small-scale referring to small, coastal boats (Parma *et al.*, 2003). Small-scale fisheries are praised because they catch roughly the same amount of fish while employing more people and consuming less fuel than industrial fisheries (Pauly, 2006; Jacquet *et al.*, 2009). While these socio-economic benefits may be important for successful fisheries policy (Pauly, 2006), sustainability also requires that the biological productive capacity of the resource be maintained in the long-term. However, common indicators of sustainability, like trends in abundance (or their proxy, catch rates) or exploitation rates (the proportion of population removed by fishing annually) (Rochet & Trenkel, 2003), tend to be unavailable for small-scale fisheries.

The assumption that small-scale fisheries can be more sustainable may be due to their seemingly benign fishing practices when compared to industrial fisheries. While only one million of the world's 51 million fishers are considered industrial fishers (Berkes *et al.*, 2001), they steer large boats capable of catching tonnes of fish at once using machines to operate nets large enough to encircle a dozen jumbo jets (Safina, 1995). Industrial fisheries also tend to be equipped with ever-improving technology, helping them locate fish and remain at sea for weeks or months at a time. On the other hand, despite having small boats

and limited technology small-scale fishers have power in numbers (50 million globally) which may allow them to mount strong pressure on marine resources. Perhaps the most important difference between industrial and small-scale fisheries is in the spatial extent of their reach. While industrial fisheries exploit from the coast far offshore to the deep seas, leaving few corners of the world's oceans untouched (Morato *et al.*, 2006; Swartz *et al.*, 2010), small-scale fishers' equipment limits them to shallower coastal areas.

In fact, the mounting body of evidence suggests that small-scale fisheries have caused species declines and even extinction. For centuries, early artisanal and later colonial exploitation has caused declines in targeted species (Jackson *et al.*, 2001; Pinnegar & Engelhard, 2008) and extinctions of marine mammals (e.g., Caribbean monk seal *Monachus tropicalis*; McClenachan & Cooper, 2008). Contemporary small-scale fishing has also caused target species declines (Sadovy *et al.*, 2003; Dulvy & Polunin, 2004; Godoy *et al.*, 2010) with cascading ecosystem effects (Ruttenberg, 2001; Campbell & Pardede, 2006), even at low levels of fishing pressure (Jennings & Polunin, 1996). In addition, one study that directly compared industrial and small-scale fishing gears (Bundy & Pauly, 2001) concluded that small-scale fishing had larger and more wide-ranging ecosystem effects than the industrial sector. Many of these examples are from coral reef fisheries, which may well be unsustainable on a global scale (e.g. Newton *et al.*, 2007).

To improve management of industrial and small-scale fishing alike, there is a call to understand the spatial patterns of fishing (Hutchings & Myers, 1994; Booth, 2000; Wilen *et al.*, 2002). Fish population declines can be masked by changes in spatial distribution of fish and/or fishing (Gillis *et al.*, 1993). Such misunderstandings may help explain some of the most severe fishery collapses (e.g. Atlantic cod, Peruvian anchovy; Hilborn & Walters,

1992). In fact, it has been suggested that the only sustainable fisheries are those unable to access a major portion of an exploited species' distribution (Pauly *et al.*, 2005).

Our poor understanding of the spatial dynamics of small-scale fishing may be improved with the use of digital technology. The few studies conducted have gathered coarse spatial fishing information by asking fishers to identify their activities on maps (e.g. Daw, 2008b) or recording coordinates of fishing grounds (Bizzarro *et al.*, 2009). Digital tracking technologies, more commonly applied to monitor animal movements or industrial fisheries (GPS tracking Bertrand *et al.*, 2007; e.g. digital tags Goldbogen *et al.*, 2008), could provide complementary detailed spatial information. This detailed spatial information could be used to calculate exploitation rate. Usually calculated as the proportion of a population caught (population size being especially illusive in data-poor small-scale fisheries; Orensanz *et al.*, 2005), exploitation rate could also be calculated spatially from fishing mortality. Fishing mortality can be estimated as the proportion of all area over which species are distributed that is swept by fishing effort during a year (i.e. the 'swept area' method) (Walters & Martell, 2004).

In this study we quantified the spatial dynamics and estimated fishing mortality rate for a small-scale, free-dive fishery in the Philippines, by combining Geographic Information Systems (GIS), time-depth recorders (TDR; previously used to study marine mammals; Goldbogen *et al.*, 2008), satellite images of habitat, and ethnographic field methods. The lantern fishery we studied operates at night: a fisher holds onto an outrigger boat as he (only men are involved thus far) swims, dives, spears, or hand-collects fish and invertebrates by the light of a lantern slung low over the bow. Quantifying where fishing occurs allowed us to 1) visualize how lantern fishers exploit the marine environment, 2) calculate fishing mortality

rate, and 3) gain new insight into the sustainability of this small-scale fishery and the appropriateness of spatial management.

## **Methods**

### **Study site, fishery, and species**

Our work is focused on the Danajon Bank in the Central Philippines, a degraded double-reef system (Fig. 4.1a). Hugging the northern and western shores of the province of Bohol, the inner and outer reefs are shallow (approximately  $\leq 10$  m) and composed of scattered and patchy coral reefs interspersed with *Sargassum* and seagrass (Meeuwig *et al.*, 2003) habitats. The reefs are in poor condition because of over-fishing and destructive activities such as blast fishing (Marcus *et al.*, 2007). One of our focal communities, Handumon (municipality of Getafe, Bohol province), has the largest concentration of lantern fishers in the region (Meeuwig *et al.*, 2003). In the other focal community of Batasan (municipality of Tubigon), fishers used to breathe compressed air through a tube while diving deep to spear fish, but turned to lantern fishing when compressor diving was banned.

At night, lantern fishers sling a shaded lantern on the front of an outrigger canoe to illuminate the benthos below (Fig. 4.1b, c). The kerosene lantern sheds a circle of light about 6 m in diameter (O'Donnell, personal observation). They walk on reefs, swim, and maneuver their boats through mazes of the ropes and poles of seaweed farms. They use a rubber-band-propelled spear to pin fish to the ocean floor or move seaweed aside to look for fish or collect seahorses or invertebrates by hand. With no modern navigational aides (they

do not use maps/charts), wearing wooden goggles (Fig 4.1c) and plywood fins, lantern fishers catch at least 20 families of reef fish (including seahorses - the focus of our long-running community-based research and management programme (Martin-Smith *et al.*, 2004; O'Donnell *et al.*, 2010) and many species of invertebrates.

### **Spatial fishing behaviour - data collection and analysis**

To quantify detailed fishing behaviour we recorded fishing trips with GPS and participant observation. From April through June of 2008, 12 fishers (7 in Handumon and 5 in Batasan) each volunteered to have one fishing trip recorded with GPS and temperature-depth-recorder (TDR). A Garmin GPSMap 76 CSx, accurate to <10 m, recorded location (latitude and longitude) and time every 30 seconds. Participants ranged from 24 to 42 years of age, covering all but the oldest fishers. We did not sample the oldest fishers because they paddle to their fishing grounds and having us accompany them would be too much of a burden. We accompanied and observed fishers on half of the trips that were recorded and for consistency the other six trips recorded were also for boats with engines. In addition we observed, but did not GPS, 10 other trips between 2006 and 2008.

To understand the depth profile of fishing, we used two methods. First we interviewed 13 Handumon expert lantern fishers about their diving limits, typical depths they go to while fishing, and constraints on dive depth (interview schedule in Appendix B). Second, we recorded fishing depths in real-time by asking fishers to wear an egg-sized (2.5 x



3.3 x 4.5 cm) TDR on a belt around their waist for an entire night's fishing. We used the Sensus Ultra by Reefnet Inc., accurate to  $\pm 0.33$  m, which recorded data every second.

We distinguished between travel and fishing behaviours by combining the GPS track with evidence of swimming from the TDR and notes from observed fishing trips. We defined *travel* as time periods when the fisher was out of the water (TDR read zero m) and the GPS recorded horizontal movement at a speed consistent with motoring or paddling. Any breaks in travel not accompanied by TDR evidence that fishers were in the water were considered travel. *Fishing* bouts were defined as time periods when the fisher was in the water (TDR read  $>0$  m). We calculated the duration and distance travelled for each behaviour. We defined a *dive* as an excursion below the surface of the water of more than one meter; because the TDRs were attached at the waist, any depth greater than one meter would result in the fisher's head being underwater. Typical dive statistics (Table 4.1) were obtained for foraging fishers, including dive duration and depth. We report means  $\pm 1$  S.D.

### **Fishing mortality rate - analysis**

To estimate Danajon Bank-wide lantern fishing mortality rate we combined distance travelled while fishing from GPS tracks with published estimates of fishing effort and shallow-water area to calculate a 'swept area' estimate (Walters & Martell, 2004). To estimate a total instantaneous rate of exploitation ( $F$ ), we multiplied catchability ( $q$ ) by annual effort ( $E$ ). Catchability is the area covered by a typical unit of fishing effort ( $a$ ; average fishing distance travelled night<sup>-1</sup> calculated from GPS tracks x 6 m width of lantern

light) divided by the total area available for lantern fishing in Danajon Bank ( $A$ ; area in waters <5 m (see results): 712 Km<sup>2</sup> (Hansen *et al.*, in press). We calculated annual effort as the product of the 200 lantern fishers in the region and the 111 to 192 nights year<sup>-1</sup> they fished (Meeuwig *et al.*, 2003). The ‘swept-area’ calculation assumes that fished areas,  $a$ , are randomly distributed across the area available for fishing ( $A$ ; analysis of fishing effort distribution in Appendix C).

If we assume any prey item in  $a$  was captured, we can then calculate the maximum fishing mortality rate each year ( $\underline{U}$ ) as  $1 - e^{-F}$ . We report  $F$  and  $U$  as a range corresponding to the range of nights year<sup>-1</sup> fished. Although these calculations are typically used to assess fishing mortality rates on individual species, we estimated  $F$  on a depth stratum because data on the spatial distribution of lantern fishery-exploited species were unavailable. .

## **Results**

### **Spatial fishing behaviour**

Lantern fishers left shore between 8.30 pm and 2.00 am and returned between 5 and 8 am (Table 4.1) spending, on average, two hours travelling to go  $14.4 \pm 6.5$  km round trip. The furthest straight-line distance fished or travelled was 11.8 km. Fishers bypassed nearby grounds to fish at more distant locations (Fig. 4.2a, b). Fishers spent an average of  $4.6 \pm 1.6$  hours covering  $3.5 \pm 1.1$  km horizontal distance while fishing. Although Batasan lantern fishers spent significantly more time fishing than those from Handumon (Wilcoxon Rank Sum:  $W = 32.5$ ,  $p=0.02$ ), there was no difference in distance covered during fishing between

the two communities ( $W = 24$ ,  $p = 0.33$ ). Once at a fishing ground, they followed ocean floor features like reef edges and shoals – or a sunken hill (Fig. 4.2c).

Integrating information from GPS tracks, diving behavior, satellite habitat maps, and participant observation we generated a three-dimensional mapping of fisher behaviour. For example, the tracks at the beginning of fisher number 3's night, show that he made two loops around a shoal (Fig. 4.2c: I, II). We linked depth and space by referring to his dive profile during these first 35 minutes. His dive profile during that time showed that the deepest dives were during the first 20 minutes, to about 4 m (Fig. 4.3 I and 4.2c I; the outer loop), followed by 15 minutes spent diving between 1 and 2 m (Fig. 4.3 II and 4.2c, II; the inner loop). This approach was then applied to the entire GPS and TDR record for each fisher. The depth logger reported '0 m' when it is above the water, confirming that the fisher exited the water and returned to his boat three times (Fig. 4.2c III, V, VI and Fig. 4.3 III), paddling to a different fishing ground. Since we joined this trip, our observations also corroborate these behaviours.

Lantern fishers exploited the shallow waters, making short dives. Two of the 11 fishers remained in such shallow water that their depth loggers never recorded depths exceeding one meter. For those who dove deeper than one meter, average dive depth according to the depth logger was  $1.9 \pm 0.68$  m and maximum dive depth for any fisher was 4.7 m (Table 4.1). Average dive duration was  $14.3 \pm 4.73$  seconds, up to a maximum of 20 seconds. There were no detectable differences in any of these parameters between Handumon and Batasan fishers (Appendix B).

A more detailed analysis of fisher's dive profiles suggests different phases or types of dives (Fig. 4.3), which participant observation confirms. For the deepest diving fisher (# 3;

Table 4.1), deeper dives were characterized by longer dive durations (Fig. 4.3 I; Spearman Rank Order Correlation:  $r = 0.73$ ,  $p < 0.001$ ,  $n = 46$ ) and longer surface intervals ( $r = 0.65$ ,  $p = 0.004$ ,  $n = 46$ ). At intermediate depths a fisher can have one hand on the boat hull and reach the seafloor with the spear in his extended other hand, for example turning over loose macroalgae with the spear (Fig. 4.3 II); he then dives closer to the bottom if he sees something to catch. Fishers may also spend extended periods in very shallow depths (Fig. 4.3, IV) to search the bottom while swimming at the surface or walking on the reef below them.

Fishers told us that the depth in their fishing grounds was generally  $5.4 \pm 1.6$  m and that although they were able to dive to  $9.4 \pm 3.4$  m, they preferred to fish shallower because deeper dives exhausted them, shortening their fishing trip (3 responses) or deeper dives wasted time (1 response). Some fishers also said they were limited to shallower depths by the amount of time they could hold their breath (5 responses) or the depth the lantern could penetrate (6 responses).

### **Fishing mortality rate**

Drawing on what we learned about lantern fishers' spatial fishing behaviour, we were able to develop an overarching assessment of the pressure they exert on the shallows they fish. While a single lantern fisher could search only a tiny fraction of the fishable depths ( $< 5$  m) of Danajon Bank (about  $0.02 \text{ km}^2$ ) per night, over the course of a year the entire fleet of 200 lantern fishers could cover between 55 and 95% of Danajon Bank ( $F$ ) for an maximum

annual fishing mortality rate ( $U$ ) of 0.4 – 0.6 (i.e. 40 – 60% of fish in waters <5 m are caught).

## **Discussion**

Although lantern fishers, like other small-scale fishers, may be technologically limited, they remain capable of exerting considerable fishing pressure on the shallow marine environment. Digital tracking combined with direct observation and fisher reports enabled this estimate of fishing mortality rate for the lantern fishery and provided novel insight into spatial fishing behaviour useful for quantifying the sustainability of small-scale fishing.

Digital tags worn by lantern fishers demonstrated that they are strictly shallow divers, which may be a strategy for continuous fishing throughout the night. Although lantern fishers reported being capable of deeper dives, the positive correlation between dive depth and post-dive surface time suggests that there is a cost associated with deeper dives. Similarly, free-diving fishers in Korea show patterns of fishing at shallower depths than the maximum depths they can reach (Hong *et al.*, 1963). In addition to gear limitations, perhaps free-diving fishers remain in the shallows because they are subject to the same physiological trade-offs that are well understood in other air-breathing diving vertebrates (Nolet, Wansink & Kruuk, 1993; Goldbogen *et al.*, 2008). That is, longer dives incur an oxygen debt and a build up of carbon dioxide requiring longer post-dive recovery times at the sea surface (Mori, 1998). Remaining at shallower depths minimizes diving costs, increasing the amount of time spent fishing (as opposed to recovering). Therefore, the shallow diving behavior exhibited

by lantern fishers may be a strategy that, as some fishers suggested, enables extended hours of fishing throughout the night.

Despite a limited dive capacity, the use of low technology gear, and highly labour intensive methods, lantern fishers exert a high rate of fishing mortality on the shallow, marine environment. The regional community of 200 lantern fishers has the potential to exploit a surprisingly large area (55 to 95 % of fishable depths in our study region) each year. Lantern fishing effort may be even higher than we estimated. Another study suggested there were 25% more lantern fishers than we assumed (Green *et al.*, 2004). If lantern fishers caught everything in their path, they would take about half of all prey items in waters <5 m ( $U = 0.4$  to  $0.6$ ). This is a maximum annual fishing mortality because certainly lantern fishers miss some prey, e.g., the most cryptic seahorses. Fishing mortality rate will not be distributed equally across fishing grounds, as revealed by an analysis of fisher logbooks over the course of a year (Appendix C). The concentration of effort in preferred grounds (a common pattern for small-scale (Pet-Soede *et al.*, 2001; Daw, 2008b) and industrial fisheries (Rijnsdorp *et al.*, 1998; Jennings *et al.*, 1999) alike) will result in higher fishing mortality rates, while other areas will remain almost untouched by lantern fishing pressure. If applied over a longer time period, the technique we developed of combining GPS, depth logger and habitat maps could be used to quantify fishing mortality rates in specific areas. These finer scale assessments are not possible with available data, but could be important for understanding the effect of fishing on sessile or sedentary species.

For individual species, the accuracy of our fishing mortality rate estimates will depend on species-specific distributions. Although the calculation we used to estimate fishing mortality rate (Walters & Martell, 2004) is typically used to assess individual species,

we estimated pressure on a depth stratum because data on the spatial distribution of species exploited by lantern fishers were unavailable. For species that exist solely within the 5 m isobath we studied, our estimate of instantaneous fishing mortality rate represents a lower bound on possible values. If a species is distributed and targeted only in certain areas or habitats within those shallows, fishing mortality rate becomes more concentrated / higher than it would be if the species extended across the entire area. In contrast, for species distributed more broadly or deeper than lantern fishers can fish, our estimate of instantaneous fishing mortality rate represents an upper bound.

Small-scale fishers may face physical or gear limitations that restrict their capacity to overexploit. Because of their shallow diving behavior, lantern fishers effectively forfeit prey in deeper waters, including potentially higher value prey or a greater density of prey. As a result, exploited species, including seahorses, may benefit from a deep sea refuge from fishing. Seahorse life history (e.g. small home range, limited mobility, complex mating system; Rowe & Hutchings, 2003; Foster & Vincent, 2004), declining catch rates (O'Donnell *et al.*, 2010), and surging global demand for use in traditional medicines and aquaria (Vincent, 1996), all suggest they are over-exploited. Yet, thousands of individual seahorses are still caught each year in our focal communities (Vincent *et al.*, 2007). The many marine reserves in the region (all in shallow water) protect some individuals. On the other hand, Outside the Philippines the tiger tail seahorse, *Hippocampus comes*, has been found in deeper waters (to at least 20 m; Kuitert, 2000) that would be inaccessible to lantern fishers. If there are inaccessible fish, they could seed the shallower, fished populations, providing some resilience to fishing.

Our quantification of spatial lantern fishing behaviour has implications for practical management of this small-scale fishery. Any conservation or fishery assessments based on fishery-dependent catch estimates may be biased toward underestimating population size because the fishers only ‘sample’ the shallows and preferred fishing grounds. In addition, communities and managers in the heavily exploited Danajon Bank should consider protecting deep-water areas that could be important sources of fish (Green *et al.*, 2004).

Although small-scale fisheries have the potential to be more sustainable than their industrial counterparts (Jacquet & Pauly, 2008; Jacquet *et al.*, 2009), our study contributes to a growing literature suggesting small-scale fisheries can have big effects on the marine environment. While small-scale fisheries may never reach the intense coverage of the seafloor of industrial fisheries (areas of the North Sea and the northeastern shelf of the United States and Canada are on average completely covered by commercial fishing several times a year: de Groot, 1984; Churchill, 1989; Messieh *et al.*, 1991; Auster *et al.*, 1996), it is easy to see how the fishing mortality rate exerted by lantern fishers would lead to depletion in the shallows when put in a regional context. The effects of a few hundred lantern fishers are magnified by the effort of the almost 30,000 other small-scale fishers using 50 types of gear on Danajon Bank (Green *et al.*, 2004).

The multiplicative power of high fishing effort to result in large cumulative catches and therefore declines (as has occurred in recreational fisheries; Cooke & Cowx, 2004) is worrisome for the sustainability of small-scale fisheries, especially in developing countries where human populations are still increasing (Pauly *et al.*, 2002). There is a clear need to temper the suggestion that small-scale fisheries are ‘our best hope for sustainable fisheries’ (Jacquet *et al.*, 2009) to reflect cumulative pressures of many small-scale fishers operating



many types of gear on many days. While the shore-based, shallow habits of small-scale fishers certainly result in more localized effects than those of industrial fisheries, the capacity of un-fished areas to replenish highly exploited areas is poorly understood (Sale *et al.*, 2005). Therefore, although small-scale fisheries remain largely un-regulated (Munro, 1996; Pauly, 2006), our study underscores the importance of developing management plans to ensure their sustainability.

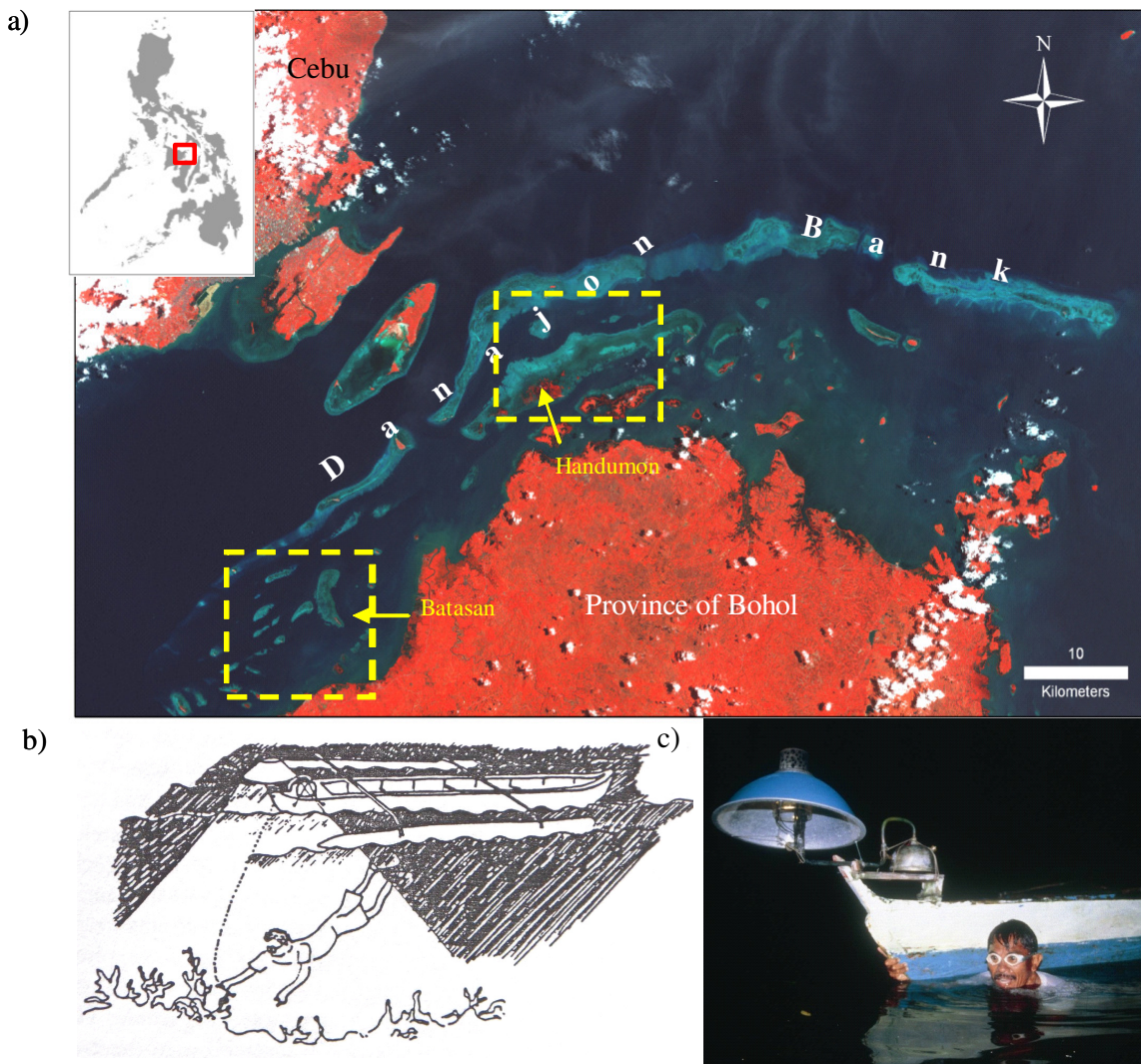
## Tables and Figures

Table 4.1. Transit, fishing, and dive statistics for each lantern fisher in Handumon and Batasan recorded by GPS and TDR.

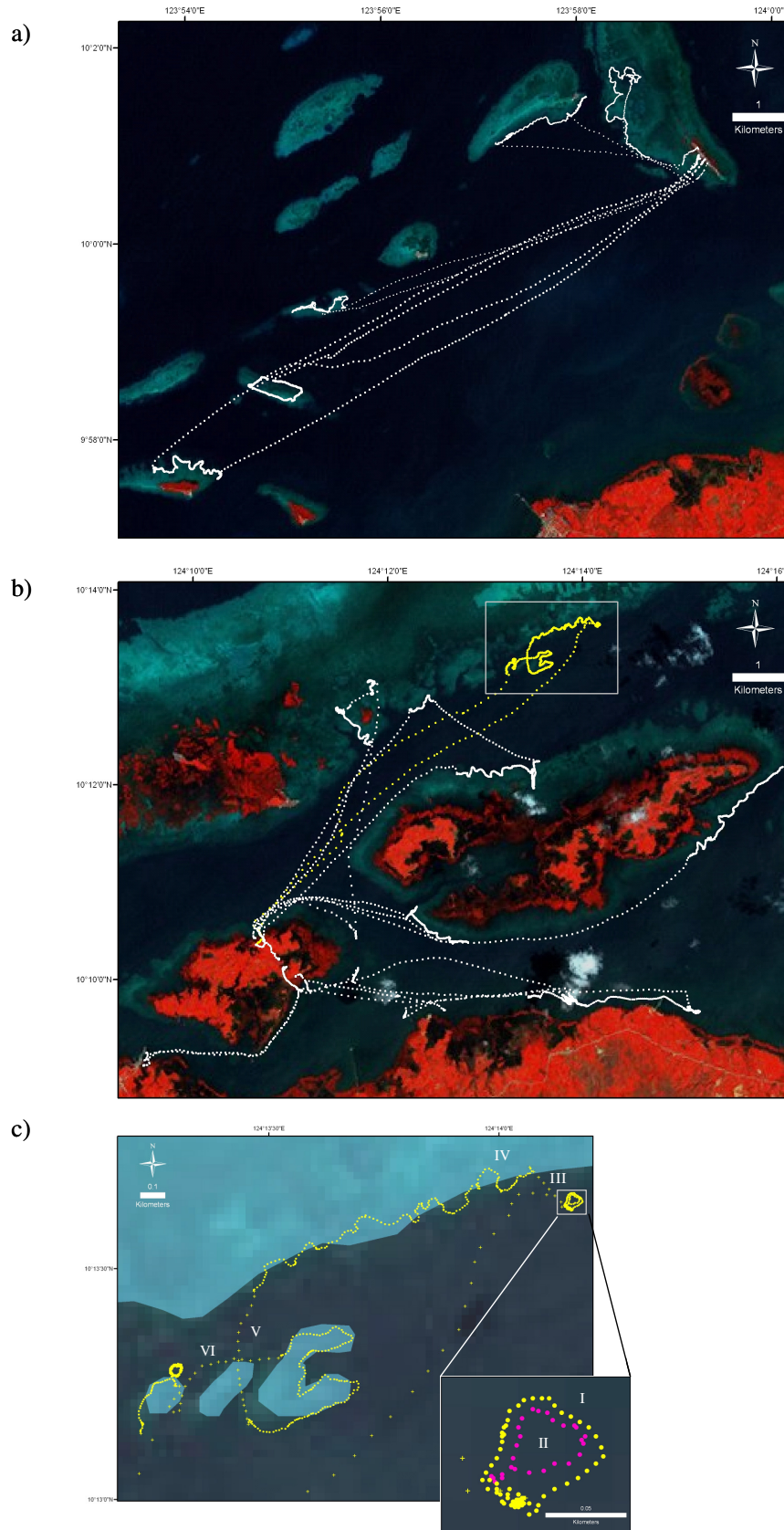
ID	Community	Start Time	End Time	Max Distance from Home (km)	Transit			Fishing			Dives				
					Total Time (hr)	Total Distance Traveled (km)	Mean Speed (km/hr)	Total Time (hr)	Total Distance Traveled (km)	Mean Speed (km/hr)	No. of Dives >1m	Max. Dive Depth (m)	Mean Dive Depth (m)	Max. Dive Duration (s)	Mean Dive Duration (s)
1	Batasan	20:27:53	5:44:53	11.8	2.4	22.9	9.5	6.9	3.3	0.5	27	1.6	1.2	17	10
2	Handumon	0:50:05	5:53:05	5.2	1.3	12.3	9.5	3.8	3.3	0.9	12	2.5	1.8	12	9
3	Handumon	23:22:05	5:54:05	8.7	1.6	17.7	11.1	4.9	4.8	1.0	47	4.1	2.4	32	19
4	Batasan	20:35:58	4:53:28	2.4	0.7	3.5	5.0	7.6	5.2	0.7	0	NA	NA	NA	NA
5	Batasan	21:49:23	6:37:23	9.4	1.7	18.4	10.8	3.9	2.6	0.7	0	NA	NA	NA	NA
6	Handumon	0:07:05	6:00:00	10.3	2.4	21.2	8.8	3.2	3.5	1.1	7	3.5	1.8	19	14
7	Handumon	23:48:53	5:32:23	6.2	1.3	12.7	9.8	4.4	4.2	1.0	2	3	2	12	9
8	Handumon	2:13:31	5:42:45	4	1	7.7	7.7	2.4	2.7	1.1	4	2.7	2.2	26	21
9	Handumon	0:26:35	8:01:05	7.6	3.5	23.0	6.6	3.4	1.9	0.6	78	4.1	2.4	31	20
10	Batasan	23:52:04	5:21:34	3.7	0.6	6.4	10.7	4.9	4.2	0.9	42	3.9	1.9	19	12
11	Handumon	1:28:34	6:30:34	5.1	1.4	11.8	8.4	3.1	1.6	0.5	23	3	1.9	24	17
12	Batasan	21:47:05	5:55:05	8.2	1.6	15.2	9.5	6.5	4.3	0.7	89	4.7	1.3	24	10

\* Did not qualify as 'diving' because spent all time in water equal to or shallower than 1 meter

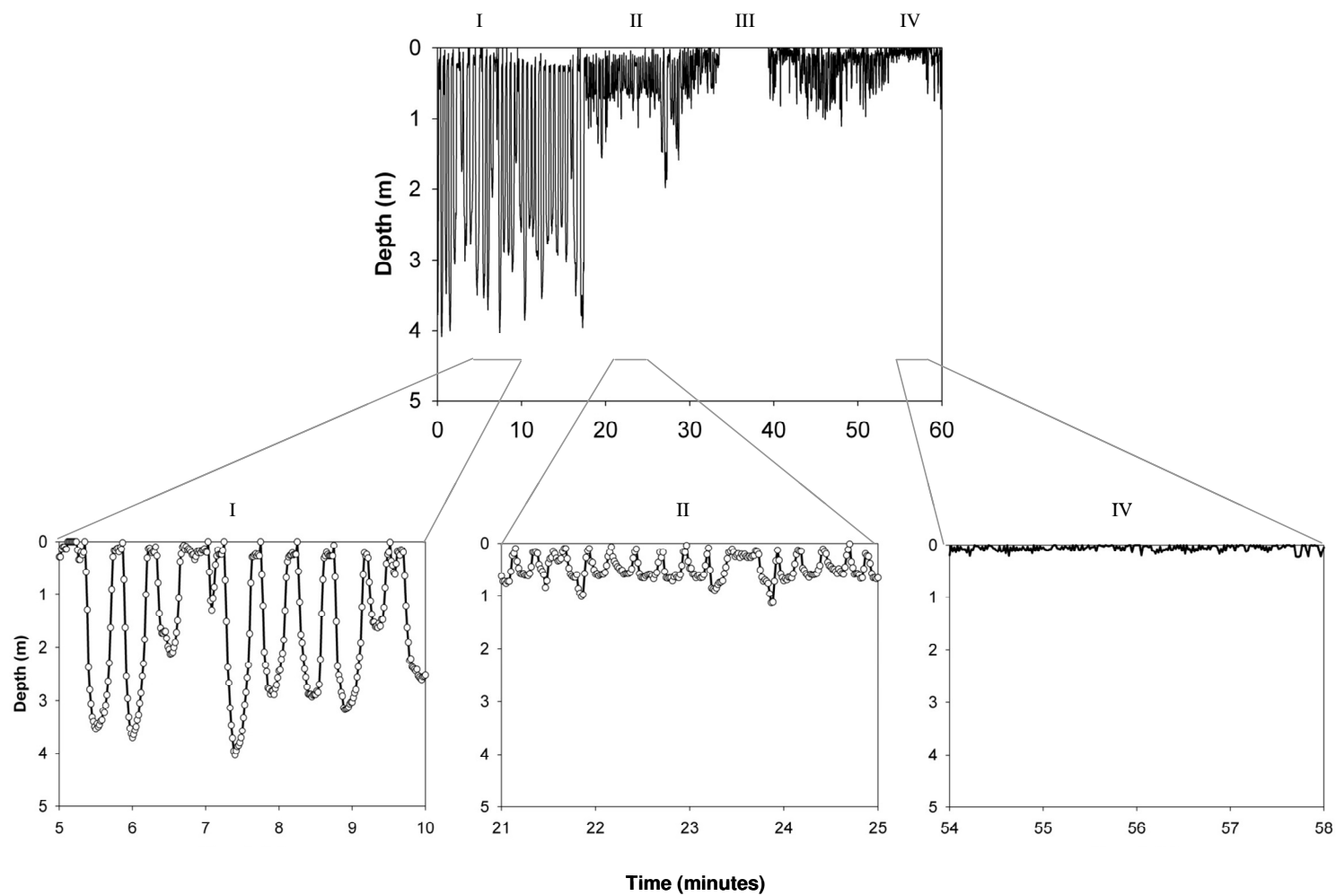
**Figure 4.1. Danajon Bank study region of the central Philippines and our focal fishing communities of Handumon and Batasan (a). Lantern fishing method (b) recreated from Vincent and Pajaro 1997 and (c) lantern fisher taking a breath at the surface.**



**Figure 4.2.** GPS tracks (white dots) of a) Batasan and b) Handumon fishers' night of fishing relative to land (terrestrial vegetation – red), shallow water reefs (light blue), and deeper water (dark blue). Section of fisher #3's GPS track (yellow in b) is shown in detail in c) where fishing (I, II, IV; filled circles) and travel (III, V, VI; crosses) behaviours are described. Inset shows inner (pink) loop and outer (yellow) loop that are linked with dive depth profile (Fig. 4.3).



**Figure 4.3.** Dive profile for fisher #3, highlighting detailed dive behaviours (I, II, IV) and out-of-water travel time (III) that can be linked with ocean floor topography (Fig. 2b, c).



## **Chapter 5: Out of reach - effects of a depth refuge on the sustainability of a small-scale fishery**

## **Introduction**

Natural refugia play an important role in protecting wildlife from overexploitation. Those species whose geographic ranges extend to inaccessible locations or even beyond where hunting is convenient or cost-effective have a spatial refuge from exploitation (e.g. Clayton *et al.*, 1997). Under certain conditions animals may disperse out of refugia, bolstering or replenishing populations in hunted areas (Allen & Sargeant, 1993; Novaro, Redford & Bodmer, 2000). Refugia can be especially important if they contain critical life stages (breeding/nursery habitat) or are the last remaining intact habitat for otherwise highly threatened species (e.g. the Amur Tiger *Panthera tigris altaica*, Novaro *et al.*, 2000).

Natural refugia from exploitation exist in the marine realm but are becoming scarce. Historically it was common for economic costs and technological limitations to create large refugia from fishing in the deep, open ocean. Contemporary studies of fisheries that cannot access an entire population (e.g. Kang *et al.*, 1998; D'Onghia *et al.*, 2009) suggest that historic natural refugia enabled fisheries to last for centuries although they were un-regulated (Walters, 2000; Pauly *et al.*, 2005). Natural marine refugia began to disappear with improvements in engine power and 'fish finding' technology that came with increased industrialization of fishing in the 1950s (Dugan & Davis, 1993; Bohnsack, 1998; Walters, 1998; Swartz *et al.*, 2010). These technological advances have not been acquired by most small-scale fisheries that predominate on tropical coral reefs, where fishing is limited to near-shore, shallow habitats (Munro, 1996).

Refugia may be particularly important for the conservation of fishes on coral reefs, where there is high biodiversity, local species have been overexploited, and tens of millions

of people continue to depend on fish for food (Pauly *et al.*, 2002; Bellwood *et al.*, 2004; Burke *et al.*, 2011). Although depth is a relatively unexplored dimension of the effects of fishing on reefs (Kahng *et al.*, 2010), two recent studies have found strong evidence for depth refugia (> 7m and > 25 m respectively) in fished reef areas, as indicated by greater commercially exploited fish species richness and relative abundance with increasing depth (Tyler *et al.*, 2009; Goetze *et al.*, 2011). These results combined with the anecdotal evidence for depth gradients reported by previous reef fishery studies suggest the potential of a geographically widespread role of depth in mediating the effect of fishing (Tyler *et al.*, 2009). The relatively shallow nature of refugia on coral reefs (starting at 10s of meters as compared to the 100s or 1000s of meters in temperate/industrial systems; e.g. Bailey *et al.*, 2009) makes them more vulnerable to encroachment by technological improvements and other disturbances (e.g. dredging or sea filling). In addition, because many coral reef species tend move to deeper waters as they get older (Brokovich *et al.*, 2007; Appeldoorn *et al.*, 2009) and older fish tend to produce more and/or better quality young (Birkeland & Dayton, 2005), deep refugia may contribute disproportionately to shallower coral reef fish populations. Little is understood, however about how these deep reef areas may be important for replenishing their shallow counterparts.

The science for understanding the roles of refugia in supporting coral reef fish populations comes largely from the burgeoning field of Marine Protected Areas (MPA) science. In theory, such refugia could support fisheries outside their boundaries if eggs or larvae dispersed out of the refuge or if increasing adult density inside the refuge caused adults to ‘spill over’ into fished areas (Dugan & Davis, 1993; Hilborn *et al.*, 2004).

Therefore, the distance, amount, and direction of fish movement between refuge and fished



areas (deemed ‘connectivity’), determines the capacity of refugia to enhance and stabilize marine fisheries. Unfortunately we have a limited understanding of the general characteristics of spatial dynamics in marine populations, especially at early life stages (Sale *et al.*, 2005; Botsford *et al.*, 2009). Therefore the effects of refugia, and any fishery benefits, remain uncertain (Hilborn *et al.*, 2004; Sale *et al.*, 2005).

Modeling can be a pragmatic tool for evaluating the importance of natural refugia to species viability and small-scale fishery sustainability. Although small-scale fisheries employ the vast majority of fishers globally (Berkes *et al.*, 2001), they are poorly understood with few data available on catches or the biology of exploited species (Orensanz *et al.*, 2005). Those on coral reefs can be especially data poor (Munro, 1996; Sadovy, 2005). When urgent decisions need to be made about threatened species, modeling allows us to combine available data and systematically incorporate uncertainties to weigh management or recovery options. For example, models allow us to incorporate uncertainty in processes like dispersal (e.g. Jones *et al.*, 2009) or density dependence (Hixon & Webster, 2002; White *et al.*, 2010) of coral reef fish, critical to consider in the design of marine reserves (e.g. Gerber *et al.*, 2005). While few modeling exercises in the marine reserve literature evaluate effects of refugia on conservation objectives (Gerber *et al.*, 2003), model outputs can easily be used to assess extinction risk using criteria like the IUCN Red List (IUCN, 2001).

This study casts the question about the importance of refugia in a slightly different light than other models of MPAs; we predict the risk (to shallow populations and catches) of beginning to fish an already established natural depth refuge. Other models of MPA effectiveness as a fishery management tool largely investigate whether establishment of a refuge can increase fishery yields despite the loss of fishing area to the refuge (Hastings &

Botsford, 1999; Gaylord *et al.*, 2005; White & Kendall, 2007). We examined evidence for a depth refuge from a small-scale, free-dive fishery in the Philippines and assessed the importance of the refuge for maintaining population size and catch, and preventing extinction of a threatened target seahorse (listed as Vulnerable on the IUCN Red List; IUCN, 2008a) given current fishing pressure. We constructed a population accounting model to quantify the effects of the refuge and evaluated the sensitivity of model outcomes to assumptions about key parameters.

## **Methods**

### **Study area, fishery, and species**

Our research is situated on Danajon Bank (Fig. 5.1), a double-reef system high in reef fish diversity, but threatened by overfishing and habitat destruction (Alcala & Russ, 2002; Green *et al.*, 2003; Carpenter & Springer, 2005). Hugging the northern and western shores of the province of Bohol, the inner and outer reefs are shallow (approximately <10 m) and composed of scattered and patchy coral reefs interspersed with Sargassum and seagrass habitats (Meeuwig *et al.*, 2003), but channels between reefs can reach 40 m (Hansen *et al.*, in press). The shallow reefs are in poor condition because of over-fishing and destructive activities such as blast fishing (Marcus *et al.*, 2007), which has spurred many communities to establish small MPAs (Marine Protected Areas – no fishing zones). The MPAs cover only a small-fraction (0.32%; Hansen *et al.* in press) of DB, are variably enforced (Samoilys *et al.*, 2007), and are only in shallow waters (~<10 m; Hansen *et al.* in press).

Our work is focused on the threatened tiger-tail seahorse (*Hippocampus comes* Cantor 1850) which is one target of multi-species extraction by small-scale lantern fishers. Two of our focal communities, Handumon (municipality of Getafe) and Guindacpan (municipality of Talibon), are known for their expertise at lantern fishing to catch seahorses. Handumon has the largest concentration of lantern fishers in the region (Meeuwig *et al.*, 2003) and catches the most seahorses, with Guindacpan a close second (Project Seahorse unpublished data). In the other two focal communities of Batasan (municipality of Tubigon) and Cataban (municipality of Talibon), fishers used to breathe compressed air through a tube while diving deep to fish (for sea cucumbers, seahorses, etc.), but turned to other gears when compressor diving was banned. Compressor fishing allowed access to species at depths beyond where most fishing activity occurs in this region and is known to have caused depletions in targeted species (Gallardo, Siar & Encena, 1995; Amar, Cheong & Cheong, 1996; Pet & Muljadi, 2001).

At night, lantern fishers sling a shaded lantern on the front of an outrigger canoe to illuminate the benthos below. This allows them to free-dive, spear, and hand-collect fish and invertebrates, but lantern fishers are limited to the upper 5 m of water by their free diving (Chapter 4). While these fishing methods may seem benign, our research suggests that lantern fishers are capable of exerting considerable exploitation rates on the shallows and could take 40 – 60% of fish each year in waters <5 m (Chapter 4).

The breath-hold nature of lantern fishing may create an important depth refuge for seahorses. Seahorse life history (e.g., small home range, limited mobility, complex mating system; Rowe & Hutchings, 2003; Foster & Vincent, 2004) and surging global demand for use in traditional medicines and aquaria (Vincent, 1996), put them at risk of over-

exploitation. With the Philippines as an important historic exporter of seahorses (Vincent, 1996), lantern fishers report that seahorse catch rates have declined dramatically (up to 93% by 1994; O'Donnell *et al.*, 2010) since they began fishing in the early 1970s. Despite these declines in the shallows and other qualitative indications of over fishing (Martin-Smith *et al.*, 2004), thousands of individual seahorses, primarily *H. comes*, are still caught by lantern fishers each year in our focal communities (Vincent *et al.*, 2007), but we do not have tallies of total regional catches or population size. Shallow MPAs certainly protect some individuals (Yasue & Vincent, in review), but a depth refuge might provide an important contribution to resilience to fishing.

*Hippocampus comes* has been best studied extensively in DB, as part of our long-running community-based research and management programme (Martin-Smith *et al.*, 2004; Vincent *et al.*, 2007; Yasué *et al.*, 2010), but only in the shallows. Overfishing and habitat damage contributed to this species' being listed as Vulnerable to extinction on the IUCN Red List (IUCN, 2008a). This fish is a medium-sized smooth, dark seahorse often with yellow-and-black-striped pattern on the tail, found around the South China Sea (Lourie *et al.*, 2004), with most sightings in the Philippines. Our studies suggest that *H. comes* lives at least 2.5 years in the wild (Morgan, 2007) and like many seahorses, it tends to maintain prolonged, monogamous pairings for reproduction (Perante *et al.*, 2002). Males give birth to live young in batches or 'broods' which can disperse 12 to 210 km (Morgan, 2007) before settling mostly in shallow sargassum habitats (Morgan & Vincent, 2007). Holdfasts grasped as anchor points by adults with their prehensile-like tails include a wider diversity of habitats (e.g. branching sponges and corals; Morgan & Vincent, 2007). Adults maintain small home ranges often no more than 1 m<sup>2</sup> (Perante *et al.*, 2002) and are patchily distributed at

extremely low densities ( $1.43 \times 10^3 \text{ m}^{-2}$ ; Morgan, 2007). Previous genetic analysis indicates that individuals from DB are likely to be part of a single interbreeding population (Casey, 1999). Although in DB *H. comes* has thus far only been studied in the shallows (< 5m), outside the Philippines *H. comes* has been found in deeper waters (to at least 20 m; Kuitert, 2000).

### **Evidence for depth refuge**

We used two approaches to evaluate whether *H. comes* inhabits waters deeper (> 5 m) than lantern fishers can fish: fisher interviews and SCUBA surveys independent of any fisheries. We relied on fisher interviews to help prioritize locations to begin searching on SCUBA.

#### ***Fisher interviews***

We consulted fishers to get a broad understanding of the depth distribution of seahorses and to identify locations for fishery independent surveys. During November and December of 2007 we conducted rapid, structured interviews with individuals in our four focal communities who fish with a variety of gears at a range of depths and catch seahorses. Since there is no official register of fishers, we consulted village leaders for initial recommendations of individuals who might be willing to be interviewed and thereafter found

respondents by word-of-mouth (Snowball sampling; Berg, 2001). For individuals who reported catching seahorses in the past 5 years, we asked for more detail about fishing ground names and locations, depths, and habitat types. Fishers generally reported depth in units of ‘dupa’ which on average converted to 1.68 m. Interviews were conducted in the local language using trained research assistants and followed an interview schedule (Table 5.1). Ultimately, we interviewed 16 fishers in Handumon, 20 in Guindacpan, 29 from Batasan, and 10 in Cataban. Numbers were determined by availability of respondents and their willingness to participate.

### ***Fishery independent survey***

We conducted an Underwater Visual Census for seahorses in waters deeper than lantern fishers can reach (from 8 to 14 m; Chapter 4) between February and June 2008. Although lantern fishers only fish to 5 m (Chapter 4), we began sampling at 8 m to be sure we were beyond their reach. We chose a total of 5 sites (Fig. 5.1) that fishers reported as having the most seahorses or the greatest vertical habitat that could serve as a seahorse holdfast. Four sites were directly offshore / down the reef slope from either current, shallow seahorse fishing grounds (Tajong tajong, Kangmungo, and Lubi) or an MPA (Lawis Toni). The fifth site (Takot pito) was in open water on a deep shoal that was a former seahorse fishing ground for compressor diving fishers.

Since *H. comes* is most easily seen at night when it rises up on holdfasts (Morgan & Lourie, 2006), we searched for seahorses between 2100 and 0300 hours. In the late afternoon

we laid two 100 m transects haphazardly through areas with vertical habitat, seeking areas most likely to hold seahorses; such bias is acceptable in light of our objectives. At night, 2 divers trained to find seahorses searched an estimated 3 m on either side of the transect and timed their swims. The exact distance searched from the transect varied due to visibility and current in order to maintain safe diving conditions. We surveyed each site only once, searching for a total of 839 minutes and an estimated 8910 m<sup>-2</sup>. When a seahorse was sighted we removed it from its holdfast to record species, sex (male, female, juvenile), head length (HL) and trunk length (TrL) (Lourie, 2003) and then returned it to its holdfast. We report all seahorse sizes as estimated standard lengths (SL in mm; Lourie, 2003) based on the relationship  $SL = 10^{\log(HL+TrL)0.881+0.5531}$  (Project Seahorse unpublished data).

## **Effects of depth refuge on sustainability**

### ***Model overview***

We developed an age-structured model to explore the impact of a depth refuge on an exploited seahorse population and associated fisheries catches. The model outcomes are affected by rates of mortality (natural and fishing), reproduction and larval dispersal, and the relative size of the refuge. We began by constructing parameters to capture the population dynamics of *H. comes*. These parameters describe how *H. comes* of different ages survive, grow, become vulnerable to fishing, and reproduce (Fig. 5.2). We predicted population size without fishing until it reached equilibrium and then ran the model with and without a depth refuge for 120 model iterations (model indices described in Table 5.2), representing 10 years.

Fished areas were in waters < 5 m and the refuge/deep extended from 5 m to the limit of *H. comes* known distribution at 20 m. Each model iteration represented a month since this timeframe provided the best trade-off between computational speed and accurately capturing the growth and mortality schedules for this species. We chose a 10-year window because it is (a) the required timeframe to assess Red List status for *H. comes* (IUCN, 2001), and (b) a relevant horizon for policy makers and local stakeholders. Once the model had resolved we used predicted population size to explore the importance of the refuge on population persistence, benefit to fisheries, and conservation assessment (IUCN Red List categorization; IUCN, 2001). The model was constructed and run in R (R Core Development Team, 2009; code available in Appendix D).

### ***Survival***

We assumed that survivorship of seahorses at age  $a$  ( $S_a$ ) declined with age in a negative exponential manner (Equation 1).  $M$  is instantaneous natural mortality rate, which is constant across age, zone (fishing vs. refuge zone), and time (Table 5.2).

$$S_a = e^{-(a-1)M} \quad \text{Equation 1}$$

### ***Growth***

We assumed that the seahorses grew according to a von Bertalanffy relationship with age (Equation 2; Morgan, 2007 ).  $L_a$  is standard length (SL in mm; Lourie, 2003) at age  $a$ ,  $L_\infty$  is the modeled asymptotic length,  $K$  is the metabolic growth coefficient (months<sup>-1</sup>), and  $a_o$  is



theoretical age at length 0 (Table 5.2). These parameter values yielded negative lengths for age 0 fish; therefore, we set all negative values to 0 (Equation 2).

$$L_a = \begin{cases} L_\infty [1 - e^{-K(a-a_0)}]; & a > 0 \\ 0; & a = 0 \end{cases} \quad \text{Equation 2}$$

### ***Reproduction***

As seahorses grew, we assumed that they became more fecund, both because of pouch capacity and because they could attract larger, more fecund females. Individual fecundity at age ( $f_a$ ) is based on male fecundity ( $mf_a$ ), because only those data are easily accessed. Males did not reach maturity until 93 mm SL, age 4 months in the model (Morgan, 2007), after which the number of embryos produced per month by a male at age  $a$  is a function of the average number of broods per month ( $b$ ; Table 5.2) and the number of embryos per brood,  $e$ , (Equation 3a) which is itself a function of  $L_a$  (Equation 3b; Table 5.2). Sex ratios of *H. comes* in our study area approximate unity (Morgan & Lourie, 2006), reflecting their monogamous mating system. Overall individual fecundity at age was calculated as  $f_a = 0.5mf_a$ .

$$mf_a = \begin{cases} be; & L_a \geq 93 \\ 0; & L_a < 93 \end{cases} \quad \text{Equation 3a}$$

$$e = \alpha + \beta L_a \quad \text{Equation 3b}$$

### ***Vulnerability to fishing***

We simulated a fishery to which seahorses recruit at 70 mm SL (Table 5.2) after which vulnerability to fishing increases linearly with length until  $L_\infty$  (Equation 4; Morgan, 2007).

$$V_a = \begin{cases} \frac{1}{1 + e^{-a}}; & L_a \geq 70 \\ 0; & L_a < 70 \end{cases} \quad \text{Equation 4}$$

We calculated catches after predicting population size at each age and time step (see below).

### ***Designate fished and refuge zones***

The population was divided into two zones  $i$ : fished (shallow, < 5 m) and refuge (deep, 5 to 20 m) (Fig. 5.2). We use this coarse, implicit spatial structure because it reflects our uncertainty about connectivity in our system. It has also been argued that such simple models of spatial structure can produce similar results to those that are more complex (White & Kendall, 2007). For our default simulations we assumed that the refuge harbored 25% of the population of seahorses (for simplicity we call this a small refuge), which is based on available information on our study species and region (Table 5.2). In the absence of better information, we assumed that (i) seahorses were evenly distributed in suitable habitat within and across fished and refuge areas and (ii) patchiness in suitable habitat was similar across the entire Bank for all depths considered here (0 to 20 m).

### ***Recruitment and larval dispersal between fished and refuge zones***

Monthly recruitment rates were based on the number of embryos ( $E_t$ ) produced by individuals in all mature age classes and both zones  $N_{tai}$  using Equation 5.

$$E_t = \sum_{a=1}^{35} \sum_{i=1}^2 f_a N_{tai} \quad \text{Equation 5}$$

Following embryo production, we simulated dispersal between the two zones. The number of embryos distributed to zone  $i$  during month  $t$  ( $E_{ti}$ ) was calculated as  $E_{ti} = p_i E_t$ , where  $p_i$  is the proportion of total embryos that moved to zone  $i$ ,  $i = 1$  or  $2$ , and  $p_1 + p_2 = 1$ . We explored the full range of possible  $p$  values, i.e., where both zones receive from 0 to 100% of the embryos. We assumed that (i) all movement between zones occurred within the first month of the juveniles' life, because most dispersal occurs during the pelagic larval stage, which only lasts for 5-10 days (Morgan, 2007), and that (ii) adult seahorses were highly sedentary with home ranges often  $< 1\text{m}^2$  (Perante *et al.*, 2002). Also based on the sedentary nature of adults, we assumed no adult migration between areas.

Embryos from one month become age 1 month fish in the next according to a Goodyear compensation ratio,  $\rho$ , (Goodyear, 1993) form of the Beverton-Holt recruitment relationship (Walters *et al.*, 2005) in Equation 6

$$N_{t+1,li} = \frac{\rho(R'_i / E'_i) E_{ti}}{1 + [(\rho - 1) / E'_i] E_{ti}} \quad \text{Equation 6}$$

where  $\rho$  is the relative improvement in survival rate as  $E$  approaches zero (Walters & Martell, 2004). Density dependence is given by  $(\rho - 1) / E'_i$  where  $E'_i$  is average, un-fished, monthly embryo production in zone  $i$  for a population with recruitment  $R'_i$ .

### ***Model initialization***

We initialized the model by seeding with 2 million fish and then allowed the total population to equilibrate without fishing. The starting population size is based on the most recent estimate of *H. comes* density in DB shallow waters ( $1.43 \times 10^3 \text{ m}^{-2}$ ; Morgan, 2007) and the area of reef on the Danajon Bank that falls within the 0 to 20 m depth range of *H. comes* ( $\sim 1420 \text{ km}^2$ ; Hansen *et al.* in press). To run this initialization, we first had to calculate the number of recruits required per month in zone  $i$  ( $R'_i$  where  $i$  = fished or refuge) needed to produce a total population of 2 million fish across both zones. We did this by solving the survival equation  $N_{1ai} = R'_i S_a$  for  $R'_i$ , assuming that population sizes in both areas are equal and setting the number of fish at  $t=1$  in zone  $i$ ,  $N_{1ai}$ , to 1 million ( $R'_i = 98,126$ ). We then used  $R'_i$  to calculate the average equilibrium monthly embryo production in zone  $i$ ,  $E'_i$ , as  $E'_i = \phi R'_i$  where  $\phi$  was calculated with Equation 7.

$$\phi = \sum_{a=1}^{a=35} f_a S_a \quad \text{Equation 7}$$

We then predicted total population size forward in time until the total population size,  $N_t$  (as described below), had been stable for 10 months. This equilibrium un-fished number of fish at each age and zone,  $N_{tai}$ , then became our starting population in fishing scenarios.

### ***Impacts of refugia on population persistence, fisheries, and conservation assessments***

Once the model was initialized with the un-fished equilibrium population we explored three scenarios of increasing fishing pressure on population size and catches that

span the range of possible extremes: 1) no fishing in either area, 2) fishing only in the shallow (deep refuge), and 3) equal fishing pressure in deep and shallow (no refuge). In scenarios with fishing, we apply a constant  $U$  of  $0.05 \text{ month}^{-1}$ , which is the upper bound of estimated current exploitation rates in the shallow waters of the Danajon Bank (Chapter 4).

For age classes 2 through 35 months, in all months after the model had been initialized with equilibrium un-fished numbers ( $t = 2, \dots, 120$ ) we predicted numbers at age and zone according to the survival equation (Equation 8).

$$N_{t+1,ai} = N_{t,a-1,i} e^{-M} (1 - V_{a-1} U_i); a = 2, \dots, 35 \quad \text{Equation 8}$$

The total population before fishing each month ( $N_t$ ) was calculated using Equation 9.

$$N_t = \sum_{a=1}^{35} \sum_{i=1}^2 N_{tai} \quad \text{Equation 9}$$

We calculated monthly catches separately for each area ( $C_{ti}$ ) using Equation 10, where  $U_i$  is the constant monthly exploitation rate (proportion of  $N_{tai}$  removed per month) for each zone, and summed these to obtain total monthly catch ( $C_t$ ).

$$C_{ti} = U_i \sum_{a=1}^{35} V_a N_{tai} \quad \text{Equation 10}$$

Once simulations were complete, we calculated the decline in population size from unfished levels and used this decline to infer an IUCN Red List status. We compared the percent population decline predicted by the model with IUCN Red List ([www.redlist.org](http://www.redlist.org)) decline Criterion A2 (IUCN, 2001) *sensu* the method described in O'Donnell et al. (2010). We evaluated differences in extinction threat and fishery catch due to presence or absence of the depth refuge for *H. comes*.

### *Unknown parameters and sensitivity analysis*

Several parameter values cannot be estimated based on available data (Table 5.2). To overcome this challenge we begin by assuming values for these parameters that reflect the information that is available. To abide by the precautionary principle (UN, 1972), however, we selected values that were slightly biased toward rendering the population less resilient to fishing pressure, hence yielding conservation management recommendations. Full justification for each of these default values is given in Table 5.2. We then explore the sensitivity of our conclusions to other values for each of these parameters.

We explored the sensitivity of model outcomes across the spectrum of assumptions about 1) the strength of density dependence, 2) the amount of dispersal of embryos into the refuge, 3) refuge size, 4) the rate of natural mortality, and 5) the rate of exploitation (Table 5.2). Refuge size refers to the proportion of the total population inside the refuge before

dispersal and fishing began  $\left( \frac{R'_2}{R'_1 + R'_2} \right)$  and in addition to the small refuge used in default

calculations, we also tested the effect of having 50 and 75 % of total initial population in the refuge, which we refer to as medium and large refuges respectively (Table 5.2). We evaluated differences in percent population change, extinction threat, and fishery catch due to presence or absence of the depth refuge.

## **Results**

### **Evidence for depth refuge**

#### ***Fisher interviews***

Fishers provided the first evidence of a depth refuge from lantern fishing and critical spatial information about likely spots to find seahorses. In total, eight types of fishing gear were used to catch seahorses in waters from ankle deep to 30 m (Fig. 5.3), far beyond the depth range of lantern fishers. The fishing grounds where most seahorses were caught were in waters shallower than 2 meters; these grounds were used by gleaners, seaweed farmers, fish weirs, and abalone divers. Crab pot fishers reported having seahorses attached to pots when they pulled them up from waters as deep as 14 m. Compressor divers, crab net and gillnet fishers reported catching seahorses across the widest range of depths (6 to 30 m). With the exception of abalone and compressor divers, seahorses were not targeted; rather they were caught as a consequence of attaching themselves to passive fishing gear and subsequently sold.

#### ***Fishery independent survey***

SCUBA surveys confirmed the presence of seahorses in waters deeper than lantern fishers can fish. In waters from 8.3 to 14 m deep we found 29 tiger tail seahorses, comprising 6 juveniles (21 %), 11 males (38%) and 12 females (41%). The smallest recorded individual measured 72 mm SL, while the largest individual was 168 mm SL. The

overall sex ratio among adults was equal (Exact Binomial test,  $p = 1$ ). We also found one female hedgehog seahorse (*Hippocampus spinosissimus*), for an overall density of 3 individuals per 1000 m<sup>2</sup>. Seahorses were found on sponge (60%) and gorgonian coral (40%) holdfasts.

## **Effects of depth refuge on sustainability**

### ***Default values***

#### *Population persistence and extinction risk*

The presence of a refuge increased the size of the total population and decreased IUCN designated risk of extinction. However, even if 25% of the population was protected, the model predicts that current shallow exploitation rate ( $U = 0.05 \text{ month}^{-1}$ ) could cause large declines in population size in just 10 years, rendering the population at least Vulnerable to extinction (Fig. 5.4a). Predictions were more concerning without the refuge, with just over 20% of the un-fished population remaining, triggering an Endangered Red Listing (Fig. 5.4b).

#### *Benefit to fisheries*

The refuge benefitted lantern fishers, who cannot fish deep waters, by increasing catches from the shallow areas. The magnitude of this improvement depended on dispersal. Maximum cumulative catch occurred when 10-20% of embryos dispersed to the refuge and



was 15% higher with than without a refuge (with: 2.34 million seahorses; without: 1.98 million seahorses; Fig. 5.5, dashed lines). In contrast, when 90% of the embryos dispersed to the refuge, cumulative catches could be as much as 52% higher with than without a refuge (with: 207,000 seahorses; without: 99,000 seahorses caught). The refuge imposed a cost to overall fishing: when a refuge was in place, fished area catch was always lower than total cumulative catch (shallow + deep) without a refuge (Fig. 5.5).

### ***Sensitivity analysis***

Model predictions regarding fished population sizes, extinction risk and fisheries benefits were most sensitive to the strength of density dependence and dispersal. Results were less sensitive to variation in refuge size, the rate of natural mortality, and exploitation rate.

#### *Population persistence and extinction risk – density dependence ( $\rho$ ) and dispersal ( $p$ )*

At current exploitation rates, a population with low density dependence ( $\rho = 1.1$ ) would need a refuge and the right amount of dispersal into the refuge to avoid collapse. With a refuge, the proportion of the population remaining increased exponentially with increased dispersal of embryos to the refuge (Fig. 5.6a, solid line). With less than about 45% dispersal to the refuge, a Critically Endangered listing would be triggered, but above 85% dispersal to the refuge, the population would not even be considered threatened. In contrast, with no

refuge the total population could not compensate for the universally heavy fishing pressure and effectively collapsed (classifying as Critically Endangered) at all levels of dispersal (Fig. 5.6b, solid line).

With increasing density dependence, the refuge became less important for sustaining populations and preventing extinction – the proportion of the population remaining with a refuge began to approximate that with no refuge (Fig. 5.6a, b). When  $\rho = 7$ , the range of extinction risk classifications with a refuge was nearly the same as without a refuge (Not threatened to Vulnerable, depending on dispersal).

#### *Population persistence – refuge size, natural mortality (M), and exploitation rate (U)*

In general, smaller refugia led to more severe declines (Fig 5.6c, d). When the amount of the initial total population protected decreased from 75% (large refuge) to 25% (small refuge), the amount of decline increased only slightly from 28% to 39%, respectively (Fig. 5.6c).

Across variation in natural mortality and exploitation rate, the refuge remained important for population persistence, and hence, decreased risk of extinction (Fig. 5.6 e – h).

#### *Benefit to fisheries*

The deep refuge provided the largest increases in shallow cumulative catch in scenarios that assumed low density dependence, a medium-sized refuge, low rates of natural mortality, and high exploitation rates. Benefits to shallow catches decreased with increasing

density dependence. With low density dependence ( $\rho = 1.1$ ), shallow cumulative catches over 10 years could be as much as 64% higher with a refuge (Fig. 5.7a solid black line) than without (Fig. 5.7a solid grey line). In contrast, at higher levels of density dependence, although the magnitude of catches was higher, the improvement in catch with the refuge decreased (when  $\rho = 7$ , max improvement in catch was 19% with refuge; Fig 5.7a dash-dotted lines).

The size of the refuge did little to change the benefit of the refuge to shallow catches, but a medium-sized refuge did provide the largest overall improvement in catches. With a medium sized refuge (50% protected) a maximum of almost 616,000 more seahorses could be caught in the shallows than without a refuge (Fig. 5.7b dashed lines), compared with a maximum benefit of 541,000 and 488,000 with small (Fig. 5.7b solid lines) and large (Fig. 5.7b dotted lines) refugia.

Low rates of natural mortality allowed for the largest improvements in catch with the refuge. At the lowest value of  $M$  ( $0.06 \text{ month}^{-1}$ ), having a refuge increased catch by a maximum of over 1.2 million seahorses (Fig. 5.7c solid lines), compared to a maximum improvement of 541,000 seahorses with the highest value of  $M$  ( $0.15 \text{ month}^{-1}$ ; Fig. 5.7c dotted lines).

Finally, across variation in exploitation rate, the refuge increased shallow, cumulative catches (Fig. 5.7d). The refuge provided the biggest benefit, when exploitation rate was highest ( $U = 0.05^{-1} \text{ month}$ , Fig. 5.7d dotted lines) increasing shallow catches by up to 541,000 over 10 years.

## **Discussion**

This study that considers how natural refugia contribute to coral reef fish population persistence and fisheries, highlights the benefits of maintaining/protecting such refugia from encroaching fishing pressure or other human-induced disturbances. Across a range of uncertainty in key parameter values, model predictions consistently suggested that the presence of a depth refuge increased *H. comes* population size, decreased extinction risk, and enhanced lantern fisher catches in the shallows. The refuge was particularly important when we assumed low density dependence, but the amount of protection and benefit to shallow catches it provided was affected both by the amount of embryos that dispersed to it and the assumed refuge size.

Fisher interviews facilitated our first documentation of *H. comes* in waters deeper than lantern fishers can reach. Fisher knowledge about where seahorses in deep waters might be located, streamlined a challenging task of locating a small, cryptic fish that occurs at low densities. The small number of deep seahorses we found during SCUBA surveys appeared to share many characteristics with shallow *H. comes* populations in terms of density and sex ratio (Perante *et al.*, 2002; Morgan & Lourie, 2006; Morgan & Vincent, 2007; Yasue & Vincent, in review). Gathering data allowing for more robust estimates of population characteristics at depth will be very challenging in coral reef systems because of the practicalities of diving in such remote locations, at depth, in search of a rare species, and in our case at night. While baited, remotely operated cameras used in other depth refuge surveys (Goetze *et al.*, 2011) may be appropriate for more mobile fish species, they are unlikely to be an effective survey tool for lower trophic level or more sedentary fish like seahorses.

In the model, the deep refuge from lantern fishing played an important role in sustaining populations and shallow lantern fishery catches of *H. comes*, even when we admitted a wide range of uncertainty in key processes and parameters. As is common for most coral reef fish and small-scale fisheries we do not know how much of the population is protected by the refuge, how connected refuge and fished populations are through dispersal, or the strength of density dependence in regulating populations (Berkes *et al.*, 2001; Sadovy, 2005; Tyler *et al.*, 2009). By examining the role of the refuge across a broad range of uncertainty in these key processes and parameters, we find that although they do influence the amount of protection provided by the refuge, they do not change the result that having a refuge in place increased population sizes and shallow catches and decreased IUCN designated risk of extinction. Only at the highest values of density dependence did population sizes and catches with and without a refuge begin to approximate each other, suggesting the refuge was less important. A population with strong density dependence can bounce back from heavy fishing because juvenile survival will increase at low adult densities (Rose *et al.*, 2001). Unfortunately, as we describe in Table 5.2, *H. comes* is unlikely to be this resistant to fishing pressure.

Prohibiting destructive and efficient gears from the depth refuge could be a pragmatic approach to preventing further collapse of *H. comes* populations and catches. None of the areas in waters deeper than 10 m are protected in DB (Hansen *et al.*, in press). Although small refugia have been established in shallow waters, the amount of area they protect (0.32% of DB; Hansen *et al.*, in press) falls short of the law requiring municipalities in the Philippines to protect 15% of their waters (Fisheries Code: RA 8550). The currently protected shallow area is also dwarfed by even our conservative estimates of the amount of

area in waters deeper than lantern fishers can reach that could support seahorses (25% of DB). Given the large area in deeper waters and their distance away from shore and the watchful eyes of communities, it may be impractical to enforce or patrol deeper areas to prevent all fishing. Mobile fishing gears (e.g. trawling) that destroy habitat and compressor fishing that has high rates of depletion (Gallardo *et al.*, 1995; Akamine, 2001; Morgan & Panes, 2004) are illegal on most of Danajon Bank. Perhaps the most pragmatic solution, therefore, would be to focus limited enforcement monies on excluding these most destructive and efficient gears from deep refugia.

A refuge alone may not be enough to sustain *H. comes* at viable population sizes. Even with a refuge, most model scenarios suggest that after just ten years of lantern fishing at current levels, *H. comes* population declines put it at risk of extinction (classified as Vulnerable to Critically Endangered). Only at very high levels of density dependence and when a large refuge was supplied with very high dispersal was the amount of population decline small enough not to trigger a threatened classification. This suggests that, in addition to a refuge, other options for reducing exploitation rates may be necessary to sustain *H. comes* at viable population sizes (as has been suggested for other species; Hilborn *et al.*, 2004). These declines may actually be underestimated because of some of the simplifying assumptions we made when constructing the model. First, the best life history data available were measured on wild, exploited populations (Morgan, 2007). Since growth and reproduction can increase in response to heavy exploitation (Heino & Godø, 2002), using these life history data to characterize equilibrium populations before fishing (and subsequent changes with fishing) may predict greater resilience than actually exists. Second, we did not incorporate any stochasticity into this model because we do not presently have an

understanding of inter-annual variation in the vital rates of *H. comes* (Morgan, 2007).

Deterministic models tend to predict that populations are more robust to exploitation, relative to stochastic models (Pitchford, Codling & Psarra, 2007), meaning that declines estimated by our model are likely to be minimum possible estimates.

Although we evaluated the importance of the refuge after just ten years, we believe that conclusions drawn after this relatively short time-frame are robust. For all parameter combinations, the ten-year time span captures most of the population change. For example, when we examined the population trajectory for the combination of parameters that would result in the most extreme declines, we found that by ten years (120 months) the population had already declined by 97%, which was within 2% of the total decline at equilibrium (99%, Appendix D). In addition, our conclusions about the importance of the refuge for sustaining shallow populations appear to apply in the long term. When we ran the model until populations came to equilibrium with fishing, conclusions about the importance of the refuge were the same as in the ten-year time frame (Appendix D). Policy-makers should consider, however, two differences in results from the equilibrium (long-term) and the ten-year (short-term) analysis. In the long-term: 1) without a refuge, the population declines more and is more at risk of extinction (e.g. at default parameter values classifies as Critically Endangered as compared to Endangered; Appendix D, Fig. D.2), and 2) with a refuge, the improvement in shallow cumulative catch is smaller (e.g. at default parameter values cumulative catches were a maximum of 49% higher as compared to 52% higher; Appendix D, Fig. D.3).

The nearly universal (across wide uncertainty in key parameters and processes) benefits of deep reef refugia suggested by our study, emphasizes the importance of protecting these areas just out of small-scale fishers' reach. Although we are only beginning to quantify

depth refugia on coral reefs, there is evidence that they may be widespread (Tyler *et al.*, 2009) and are an important component for sustainable fisheries (e.g. Karpov *et al.*, 1998; D’Onghia *et al.*, 2005). If other refugia support population persistence and shallow catches as broadly as in our study, protecting deep (>10 m) reef areas could help safeguard the food supply shallow reefs provide for millions of the world’s most poor (Moberg & Folke, 1999). Furthermore, if refugia in other locations are as large as we estimate they could be on Danajon Bank, protecting them could help reach global marine protected area goals (10% by 2020; CBD, 2006). More research is needed to quantify the contribution that un-fished deep reef areas make to the shallows for other species. Our work provides pragmatic methods for identifying refugia and evaluating their local importance.



## **Tables and Figures**

**Table 5.1. Interview schedule**

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**Biographical information:** name, gender, age

**Seahorse catch:**

In the past 5 years, have you ever caught seahorses?

What kind of fishing gear were you using when you caught seahorses?

What are the names of the fishing grounds where you have caught seahorses?

Can you describe each fishing ground (location, depth, habitat)?

How often do you visit this fishing ground?

How often do you catch seahorses at this fishing ground?

What do you do with the seahorses you catch?

**Other:** How big is a dupa?

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**Table 5.2. Symbols, definitions, values (and range used in sensitivity analysis), sources of indices and parameters used in model, and justification for default value choices.**

Category	Parameter	Definition	Units	Value	Source	Justification
<b>Indices</b>	$a$	age	months	1...35	-	See text
	$i$	depth zone	-	1: shallow, 2: deep	-	See text
	$n$	maximum model iterations	months	120	-	See text
	$t$	month	-	1...n	-	-
<b>Survival</b>	$M$	natural mortality	proportion month <sup>-1</sup>	0.15 (0.06 – 0.15 month <sup>-1</sup> )	Martin-Smith et al. 2004, Morgan 2007	Chose highest estimated value
	$S_a$	survivorship – the probability of surviving from age 1 to age $a$	a probability	$S_1 = 1, S_{35} = 0.006$ , Equation 1	Walters and Martell 2004	Captures the cumulative effect of natural mortality on fish as they age
<b>Growth</b>	$a_0$	Von Bertalanffy theoretical age at which length is zero	months	0.36	Morgan 2007	Growth parameters estimated using mark-recapture data, fishery landings, and aquarium-reared juveniles of known age.
	$K$	Von Bertalanffy metabolic growth coefficient	-	0.24 month <sup>-1</sup>	Morgan 2007	See above
	$L_a$	length at age $a$	mm	$L_1 = 23.8$ mm, $L_{35} = 167$ mm, Equation 2	Morgan 2007	See above
	$L_\infty$	Von Bertalanffy length as $a$ goes to infinity	mm	167 mm	Morgan 2007	See above
<b>Reproduction / Recruitment</b>	$b$	brood number – number of broods each male has in a month	broods	1.79 broods male <sup>-1</sup> month <sup>-1</sup>	Morgan 2007	Calculated from the average reproductive duration of 17 days (Morgan 2007).
	$e$	number of embryos per brood	embryos	Ranges from 303 to 522 embryos male <sup>-1</sup> , Equation 3b	Morgan 2007	Fecundity increases linearly as a function of body size. The relationship between size and fecundity was derived from an analysis of male brood size and lengths measured for 95 captured <i>H. Comes</i> .
	$E'_i$	total un-fished monthly embryo production in zone $i$ for a population with recruitment $R'_i$	embryos	$\phi R'_i$	Walters and Martell 2004	See text
	$E_t$	total embryo production in month $t$ (summed across depth zones $i$ )	embryos	Equation 5	-	See text
	$E_{it}$	The number of embryos distributed to zone $i$ during month $t$	embryos	$p_i E_t$	-	See text
	$f_a$	fecundity at age $a$ – monthly embryo production from one age $a$ fish	embryos month <sup>-1</sup>	$f_{1 \text{ to } 3} = 0, f_4 = 270, f_{35} = 466$ , Equation = $0.5mf_a$	Morgan 2007	The relationship between size and fecundity was derived from an analysis of male brood size and lengths measured for 95 captured <i>H. Comes</i> .
	$mf_a$	male fecundity at age $a$	embryos month <sup>-1</sup>	$f_{1 \text{ to } 3} = 0, f_4 = 540, f_{35} = 932$ , Equation 3a	Morgan 2007	Fecundity increases linearly as a function of body size. The relationship between size and fecundity was derived from an analysis of male brood size and lengths measured for 95 captured <i>H. Comes</i> .
	$R'_1$	total un-fished age-1 recruitment in zone 1 - fished/shallow	fish	147189 fish, 75% of population protected [49063 (25%), 98126 (50%) protected]	-	Since the portion of the population in the refuge at un-fished equilibrium is unknown, we assumed that it was 25% of the total population. Certainly less than 50% of the total population currently resides in the deep, so we assume this was also true before fishing. Shallow and deep areas may have similar seahorse densities (see results and discussion) and the amount of area in the shallow (<5 m) and deep (5 to 20 m) is roughly equal (Hansen et al. in press), but there is less available vertical habitat at depth with vast areas as open sand or mud flats (O'Donnell personal observation). Since seahorses show strong preference for distinctive or artificial structures (Foster & Vincent 2004) and we only found individuals associated with vertical habitat, we assumed that the total number of individuals in the deep was less than in the shallow, choosing 25% as a best guess.
	$R'_2$	total un-fished age-1 recruitment in zone 2 - deep/refuge	fish	49063 fish, 25% of population protected [147189 (75%), 98126 (50%) protected]	-	See above

Table 5.2. Continued

Category	Parameter	Definition		Value	Source	Justification
<b>Reproduction / Recruitment</b>	$p_i$	proportion of total embryos that settle or move to zone $i$ , by age $a=1$ , where $p_1 + p_2 = 1$	-	0-1.0	-	This is the full range of possibilities.
	$\alpha$	intercept of linear relationship between fecundity and size	-	-2.69	Morgan 2007	The relationship between size and fecundity was derived from an analysis of male brood size and lengths measured for 95 captured <i>H. Comes</i> .
	$\beta$	slope of linear relationship between fecundity and size	-	3.14	Morgan 2007	The relationship between size and fecundity was derived from an analysis of male brood size and lengths measured for 95 captured <i>H. Comes</i> .
	$\phi$	Lifetime fecundity of one fish - the sum over ages of fecundity $f_a$ weighted by survivorship $S_a$	embryos	1730 embryos, Equation 7	Walters and Martell, 2004	
	$\rho$	Goodyear recruitment compensation ratio	-	1.5 (1.1 – 7)	Goodyear 1993, Myers et al. 1999	<i>H. comes</i> has 'weak' density dependence ( $\rho = 1.5$ ) because, according to a version of life-history theory (Winemiller & Rose 1992), this species groups most closely with equilibrium strategists (Morgan 2007) which have relatively low fecundity, and, despite high early life stage survival, should result in low maximum reproductive rates at low population sizes (Rose <i>et al.</i> 2001).
<b>Population size</b>	$N_{tai}$	population at the start of month $t$ (before fishing) for age $a$ in zone $i$	fish	Equation 8	-	See text
	$N_t$	total population at the start of month $t$ (before fishing), summed across ages $a$ and depth zones $i$	fish	Equation 9	-	See text
<b>Fishing</b>	$C_{ti}$	total catch in month $t$ and zone $i$ (summed across ages $a$ )	fish	Equation 10	-	See text
	$C_t$	total catch in month $t$ (summed across ages $a$ and depth zones $i$ )	fish	$\sum_{i=1}^2 C_{ti}$	-	See text
	$U_1$	monthly exploitation rate in zone 1	proportion of N removed	0.05 (0 – 0.05 month <sup>-1</sup> )	Chapter 4	I chose the upper bound of my estimate of $U$ from Chapter 4 as the default value and evaluated the sensitivity of the model to the full range of $U$ that I estimated.
	$U_2$	monthly exploitation rate in zone 2	proportion of N removed	0.05 (0 – 0.05 month <sup>-1</sup> )	Chapter 4	See above
	$V_a$	vulnerability to fishing at age $a$	proportion of Na vulnerable	$V_1 \text{ and } 2 = 0, V_3 = 0.95, V_{17 \text{ to } 35} = 1$ , Equation 4	-	Catch landings data from 1996-1998 and 2002-2004 used in length-converted catch curve to show that extraction of <i>H. comes</i> began at ca. 70 mm SL, with full exploitation occurring above 150 mm SL (Morgan 2007).

**Figure 5.1.** Danajon Bank study region of the central Philippines, our four focal communities where interviews were conducted, and UVC survey sites (red stars)

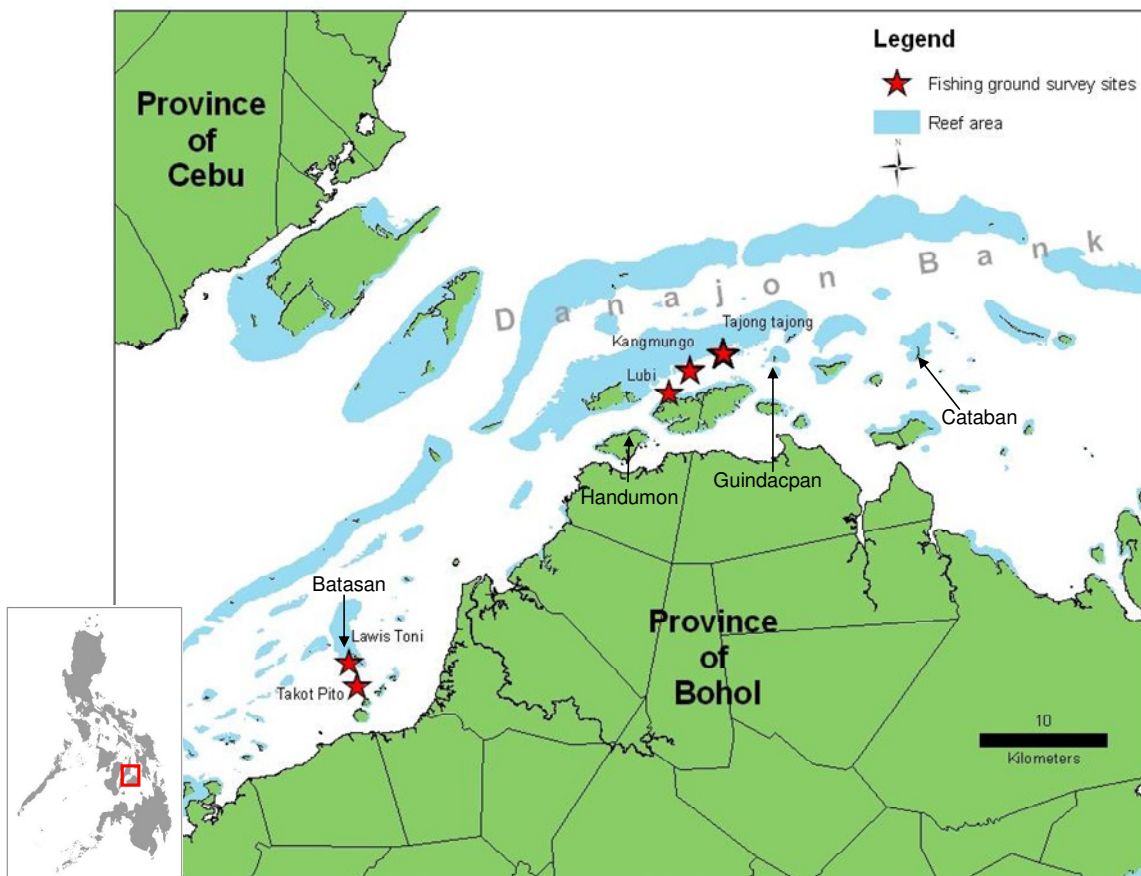
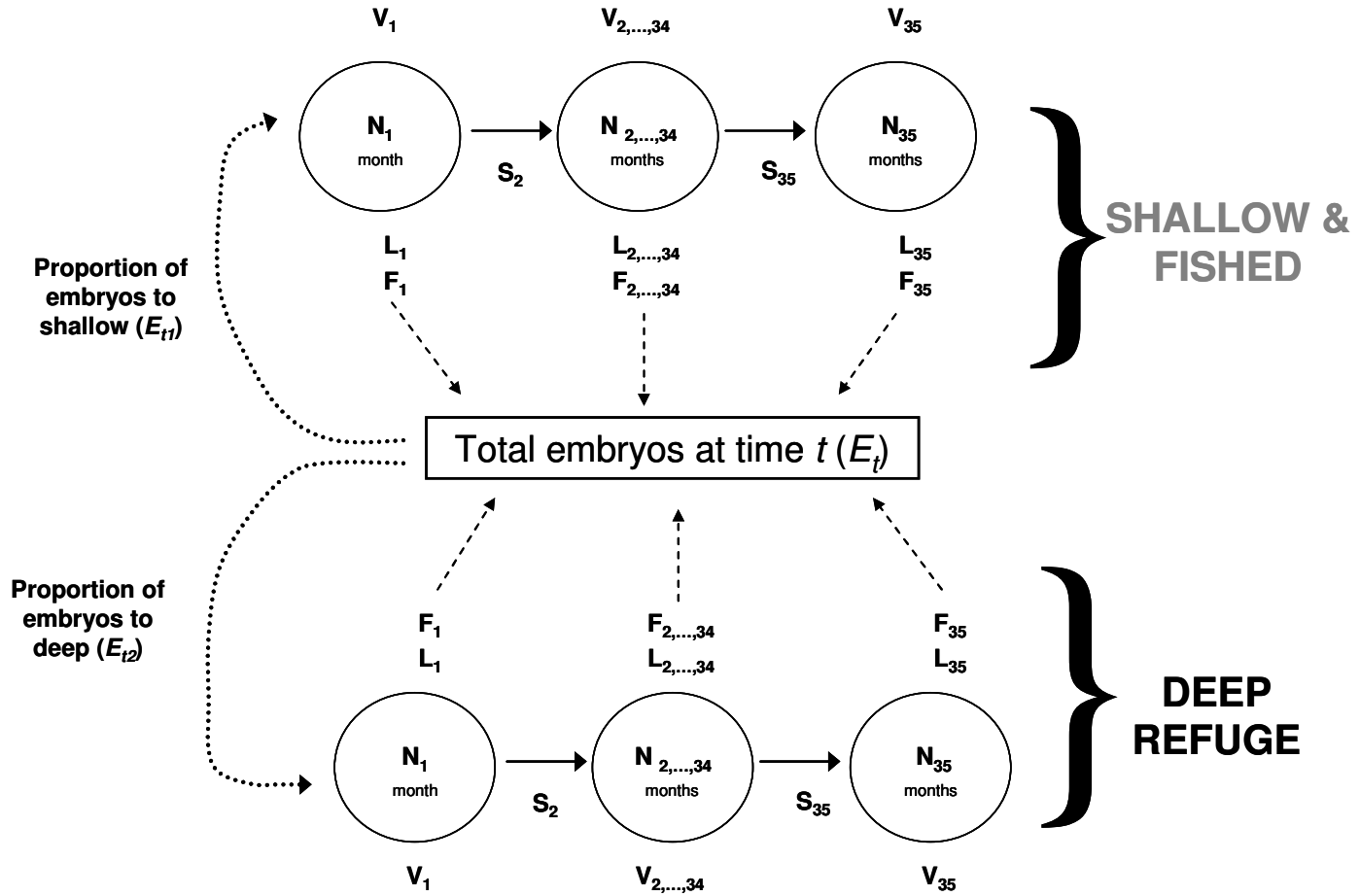
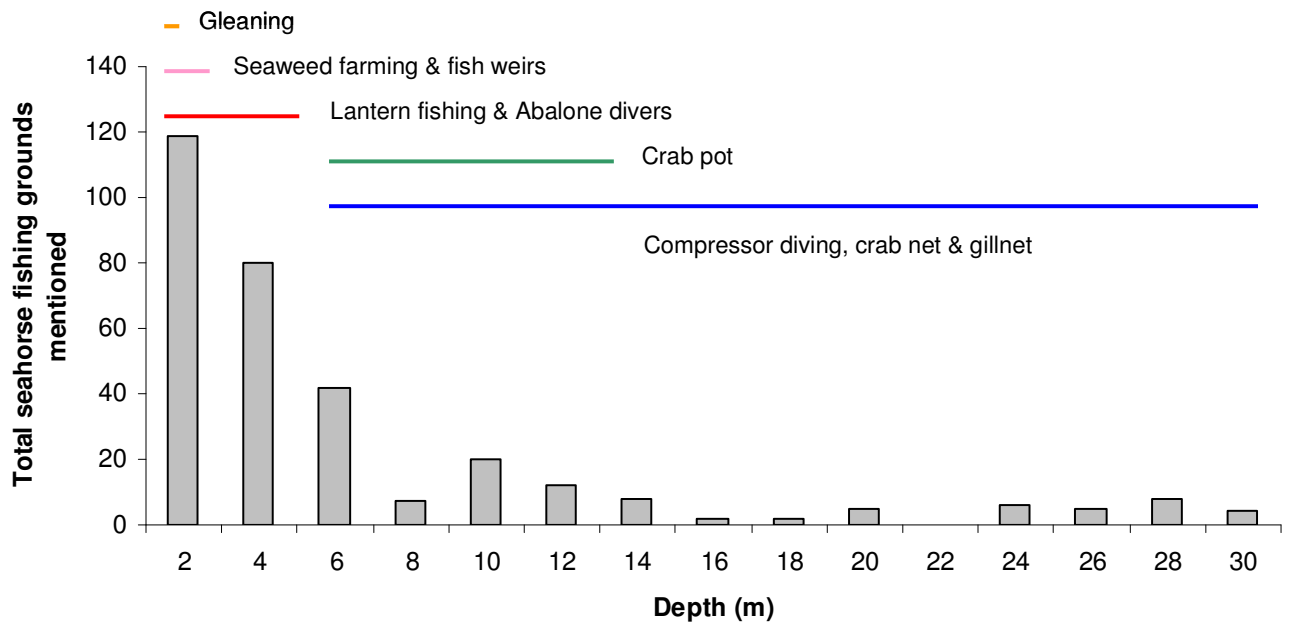


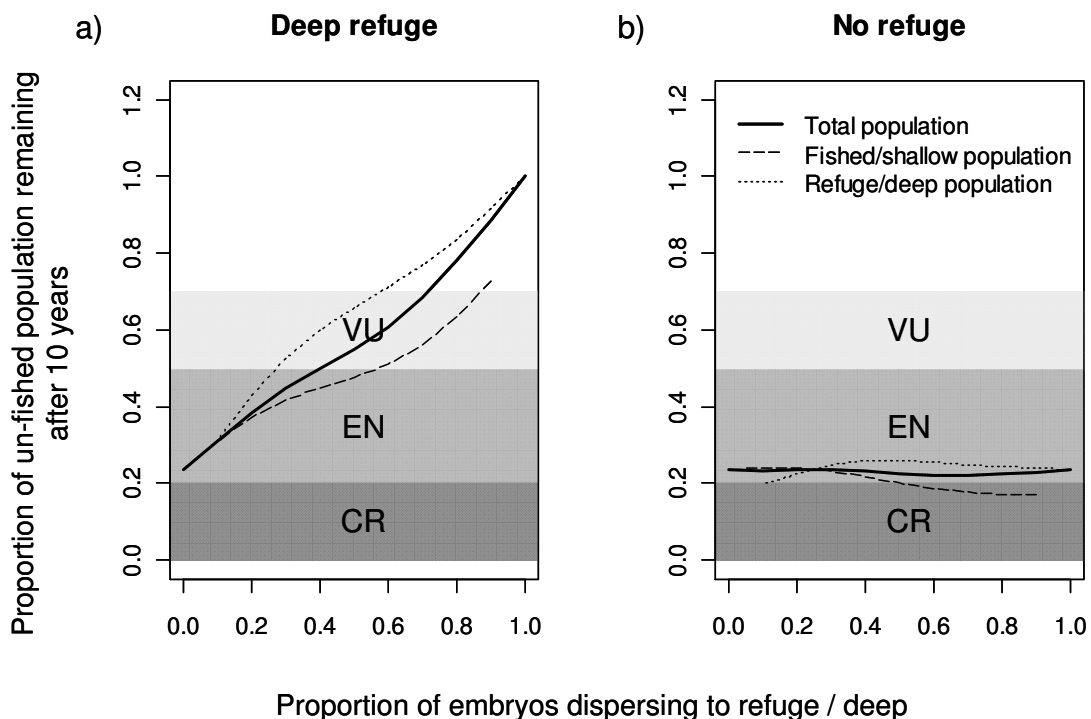
Figure 5.2. Conceptual representation of model where a meta-population (a shallow/fished and a deep/refuge population) is linked through embryo dispersal. Numbers of fish ( $N$ ), survivorship ( $S$ ), length ( $L$ ), fecundity ( $F$ ), and vulnerability to fishing ( $V$ ) change over age (1-35 month - subscripts).



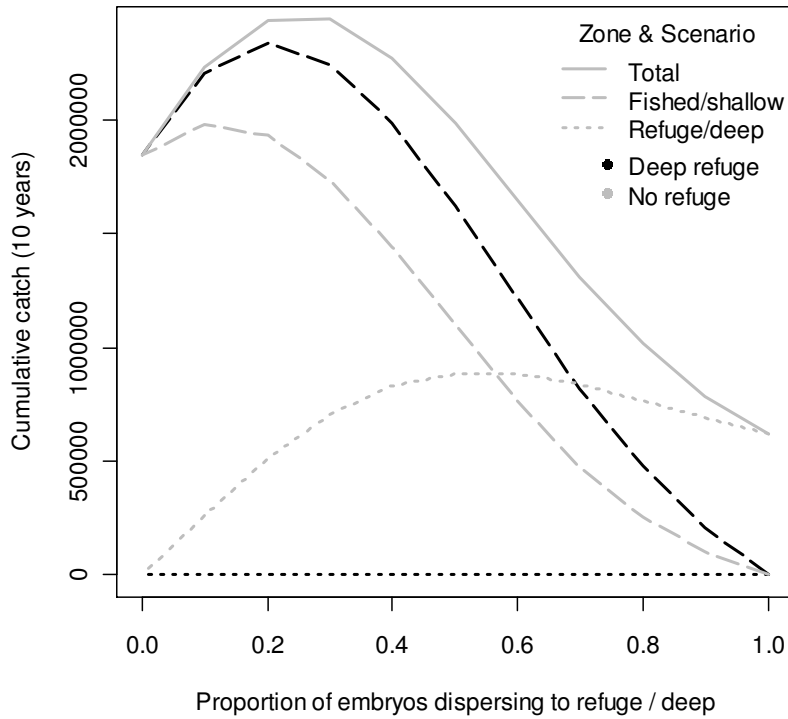
**Figure 5.3.** Frequency of interview mentions of fishing grounds where seahorses have been caught at particular depths and the range of depths fished by gear.



**Figure 5.4.** Proportion of total (solid line), shallow (dashed line), and deep (dotted line) *H. comes* populations remaining after 10 years at default parameter values and across a range of uncertainty about the proportion of embryos that disperse to the deep. We compare two scenarios: 1) a small, deep refuge where 25% of the un-fished equilibrium population was assigned to the refuge and 75% were fished at current exploitation rate (a), and 2) no refuge where 100% of the un-fished equilibrium population was fished at current exploitation rate (b).

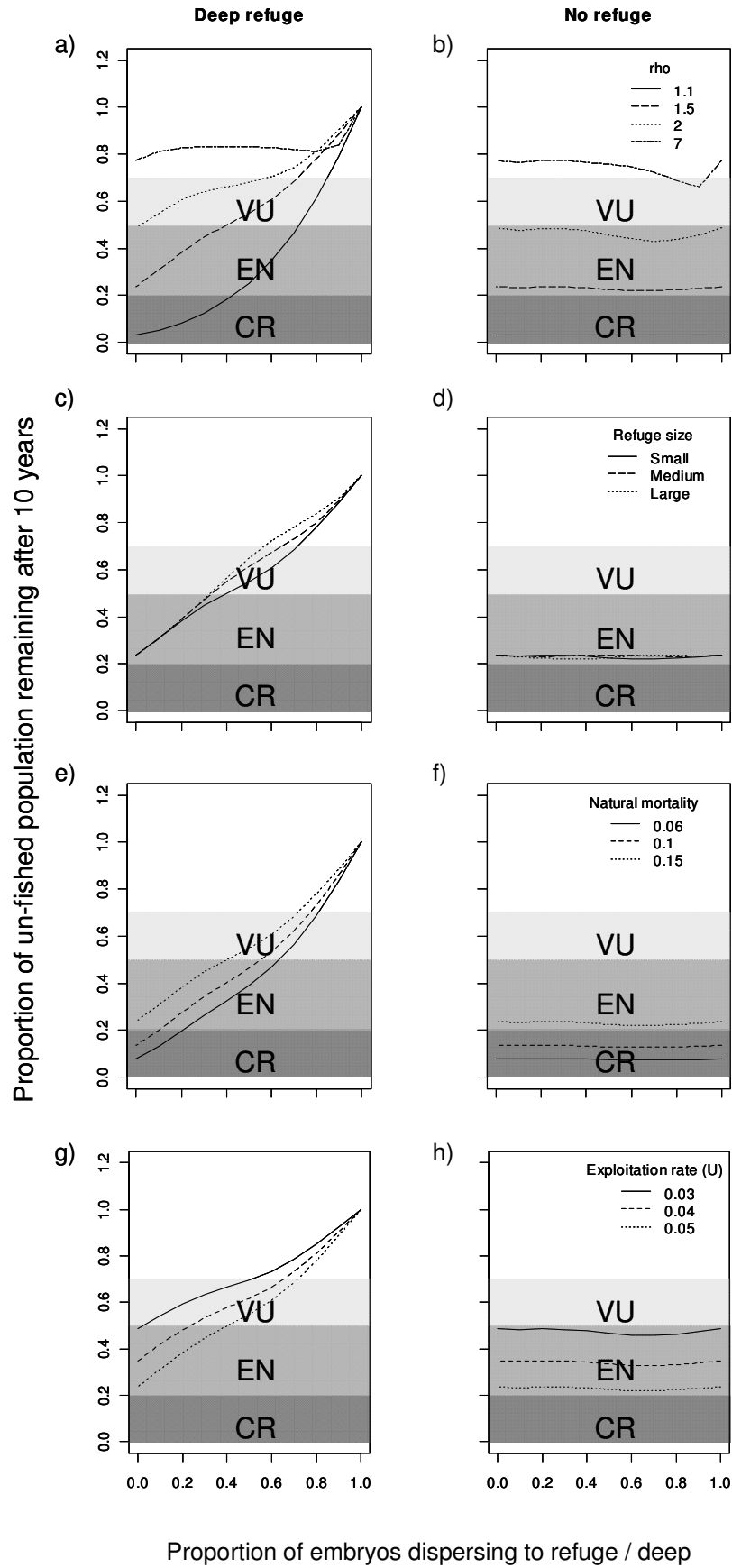


**Figure 5.5.** Comparison of cumulative catches of *H. comes* after 10 years from the shallow (dashed), deep (dotted), or total (shallow+deep; solid line) zones. We compare two scenarios: 1) a deep refuge (black lines) and 2) no refuge (grey lines).



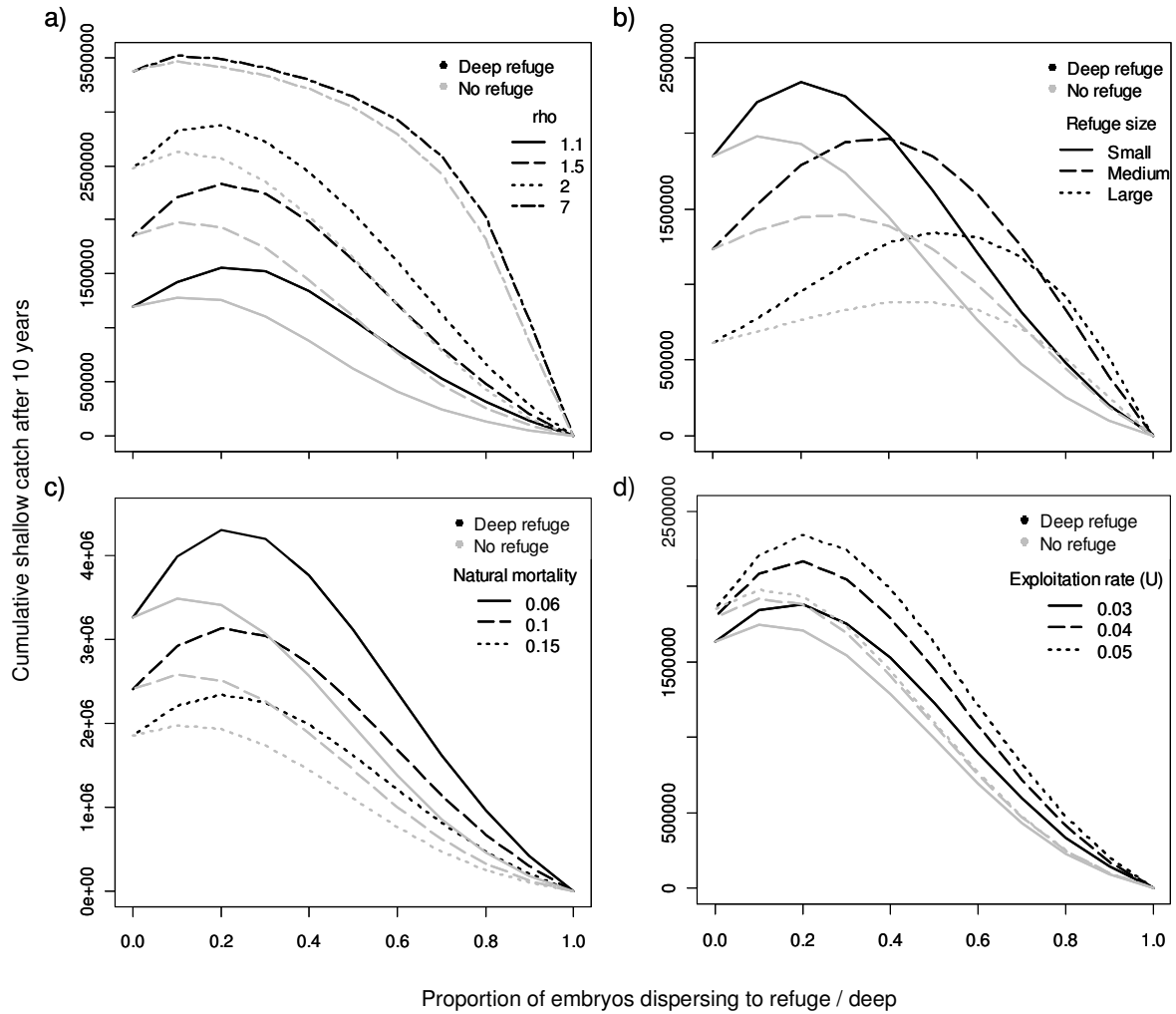
**Figure 5.6.** Proportion of total *H. comes* population remaining after 10 years given a range of assumptions about strength of density dependence ( $\rho$ , a, b), refuge size (c, d), natural mortality (e, f), and exploitation rate (g, h) and a range of uncertainty about the proportion of embryos that disperse to the deep. For each parameter we compare two scenarios: 1) a small, deep refuge where 25% of the unfished equilibrium population was assigned to the refuge and 75% were fished at current exploitation rate (left column), and 2) no refuge where 100% of the unfished equilibrium population was fished at current exploitation rate (right column).

Figure 5.6.





**Figure 5.7.** Comparison of cumulative catches of *H. comes* after 10 years given a range of assumptions about (a) density dependence ( $\rho$ ), (b) refuge size, (c) natural mortality, and (d) exploitation rate. We compare two scenarios: 1) a deep refuge (black lines) and 2) no refuge (grey lines).



## **Chapter 6: General discussion and conclusions**

## **Summary of thesis and status of research questions**

By successfully applying pragmatic approaches I have cast new light on how we might reconcile fisheries and conservation for species threatened by small-scale fishing, with a particular emphasis on seahorses. Although the global trade for seahorses has operated for decades, I provided the first historic reconstruction of a seahorse fishery. More generally, I presented evidence that assumptions made when analyzing local knowledge affect conservation assessments for species at risk. Comparisons I made between data elicited from resource users and data obtained using scientific approaches will help to develop robust approaches for incorporating fisher knowledge into quantitative assessments. I next conducted a novel analysis of spatial fishing behaviour that illustrated the capacity of small-scale fisheries to exert considerable pressure on marine systems, raising questions about their inherent sustainability. I also contributed to marine conservation planning by quantifying the importance of deep reef refugia for sustaining shallow, overexploited reef fishes and fisheries.

In this concluding chapter I summarize my collective research in applying pragmatic approaches and indicators to conservation problems and then highlight advances made toward reconciling fisheries and conservation. In addition, I discuss implications for future research both for my case study system and more broadly for threatened species and small-scale, data poor fisheries. I tapped into unconventional data sources in each of my chapters for rapid, cost-effective, information about species abundance and exploitation. Local knowledge featured prominently in Chapters 2, 3, and 5, and I directly evaluated the reliability of local knowledge in Chapter 3. I considered the behaviour of fishers in Chapter

4 and explicitly evaluated the effects of uncertainty on assessments in Chapters 2 and 5. Despite incomplete knowledge, I reached conclusions about extinction risk and fishery status in all chapters and made recommendations for action in Chapter 5. The precautionary principle featured prominently in my analysis in Chapter 5.

Throughout my thesis, I combined indicators typically used in fisheries and conservation to quantify the approaches and to track the status of my case study fishery and threatened seahorse population. I quantified trends in catch rate (CPUE) in Chapters 2 and 3 and conducted IUCN Red Listing assessments in Chapters 2 and 5. I calculated exploitation rate in Chapter 4 and quantified the extent and potential contribution of a natural refuge from fishing in Chapter 5.

I now describe how integrated application of five sets of approaches and indicators (as outlined in chapter 1) has allowed me to address my research questions with considerable success.

### **How has CPUE changed over time according to fisher perception and what does that tell us about extinction risk?**

In the first reconstruction of a seahorse fishery (Chapter 2), I found that fishers perceived severe historic declines in CPUE that were not apparent in more recent logbook trends. Furthermore, the historic baseline set by interviews suggested that relying on recent decades of data alone may have underestimated extinction risk for the study species, and others that have been exploited historically. The current IUCN Red Listing of Vulnerable for

*H. comes* may underestimate the actual threat level, especially if the biggest declines occurred before 1990.

### **How do assumptions about the accuracy of fisher recall affect the assessment of extinction risk?**

Also in Chapter 2 I developed a novel approach to correct for retrospective bias and explored its effects on conservation assessments. While local knowledge is increasingly being incorporated into conservation and fishery assessments, little is yet understood about its biases and there are no formal rules for how it should be analyzed or interpreted. For example, retrospective bias (how accurately fishers report past events) has been acknowledged in the literature (Neis *et al.*, 1999b; Ainsworth *et al.*, 2008), but there have been no attempts to quantify it, correct for it, or evaluate its effects on conservation assessments that use local knowledge. As is typical of small-scale fisheries, there were no multiple datasets that overlapped in time, so it was not possible to test the reliability of interviews by direct comparison with logbooks. Instead I developed simple calculations of CPUE for logbooks that might account for recall bias in interviews, allowing me to link interview and logbook data into one time series.

My calculations showed that the relationship between perceived historic declines in interviews and recent logbook trends depended on assumptions I made about fisher recall bias. For example, assuming fisher historic interviews were accurate resulted in the most severe declines and the most worrying extinction risk classifications. To my knowledge, this

is the first study to show that assumptions made when analyzing local knowledge affect species conservation assessments. These findings highlight the importance of avoiding overly simplistic assumptions when employing resource user knowledge.

### **How comparable are three methods of quantifying CPUE (fisher interviews, fisher logbooks and ‘scientific’ catch landings)?**

My results from Chapter 3 suggest that interviews or logbooks may be a reasonable proxy for more costly catch landings or underwater surveys (Johannes, 1998) when seeking information about trends, particularly given the need for rapid, low cost, and reliable assessment methods to detect trends in the abundance and use of wildlife (Danielsen *et al.*, 2005). When compared directly, fisher and scientific knowledge have been found either to agree (Neis *et al.*, 1999b; Begossi, 2008; Lozano-Montes *et al.*, 2008) or to disagree (Otero *et al.*, 2005; Daw, 2008a). Given that my work in Chapter 2 showed that assumptions made about biases in fisher knowledge can affect conservation assessments, there is a continuing need to understand the relationships between user-generated knowledge and other datasets (Gavin & Anderson, 2005; Gilchrist *et al.*, 2005; White *et al.*, 2005; Gagnon & Berteaux, 2009). I found that fisher interviews and logbooks provided estimates of CPUE trends consistent with more formally collected catch landings.

My results in this chapter also highlight two important limitations of local knowledge. First, interviews may not allow inferences about the absolute value of catch rates. When compared directly, interviews yielded higher and more variable estimates than other

methods, even than logbooks which were also fisher-derived. While this result may vary from fishery to fishery (other studies have found agreement e.g. Daw, 2008a), it highlights the importance of comparing data-collection methods when setting recovery targets or catch limits that are sensitive to the magnitude of reported values. Second, it may not be possible to quantify how fisher reports relate to scientifically collected data at a scale (the fishery) relevant to management/conservation because of variation among individual respondents. Having retrospective interview and landings data from the same individual fishers also allowed me a rare insight into the differences that may exist at the individual level. When I plotted interview responses against landings recorded for the same fisher in the same year, I hoped to find a consistent relationship (correlation) across fishers that might allow me to calculate a ‘correction factor’ that would adjust values provided in interviews to put them in the same scale as landings reports. In contrast to other studies (Anadón *et al.*, 2009; Rist *et al.*, 2010), I found that interview responses were not correlated with recorded landings, making it impossible to calculate a correction factor.

**What are the spatial dynamics of small-scale fishing? And what does an estimate of exploitation rate say about the sustainability of small-scale fishing?**

Using a novel method of digitally tracking small-scale fishers at work, my results from Chapter 4 suggest that, despite technological limitations, small-scale fishing can be so intense as to challenge arguments that these fisheries are necessarily a sustainable alternative to their industrial counterparts (cf. Jacquet & Pauly, 2008; Jacquet *et al.*, 2009). Information

about the spatial behaviour of small-scale fishing is largely unavailable (Salas & Gaertner, 2004). The few studies conducted have gathered coarse spatial fishing information by asking fishers to identify their activities on maps (e.g. Daw, 2008b) or recording coordinates of fishing grounds (Bizzarro *et al.*, 2009). Detailed spatial information from GPS tracks and depth tags revealed that lantern fishers were strictly shallow divers and provided data to calculate exploitation rate. My results reveal the surprising capacity of small-scale fisheries to exert considerable pressure on the fishes within the reach of their limited-technology gear. Because the reach of small-scale fishers does not extend as deep or far offshore as industrial fisheries, there may be opportunity for exploited species to survive in natural refugia.

**What role does a depth refuge play in preventing short-term risk of species extinction and in supporting shallow catches given current exploitation rates?**

My findings in Chapter 5 highlight that although deep reef refugia are not a panacea, they can benefit both shallow reef fishes and fisheries. Interviews with fishers facilitated the first documentation of *H. comes* in waters deeper than lantern fishers can reach. While deep reef refugia have been identified as potentially important and under-studied (Tyler *et al.*, 2009; Kahng *et al.*, 2010; Goetze *et al.*, 2011), this is the first consideration of how depth refugia might contribute to coral reef fish population persistence and fisheries. Combining fisheries modeling tools with IUCN Red List conservation assessments revealed that across a range of uncertainty in key parameter values, the presence of a depth refuge increased *H. comes* population size, decreased extinction risk, and enhanced lantern fisher catches in the



shallows, given current exploitation rate (Chapter 4). Even with a refuge, however, model scenarios predict worrying declines in *H. comes* populations, suggesting that a depth refuge alone may not be enough to sustain viable populations of species exploited by small-scale fisheries.

### **Reconciling small-scale fishing and conservation**

Having addressed my five original research questions, I now reflect on how my findings might make a material difference for seahorses, small-scale fisheries, and society. I also highlight how my work begins to reconcile fisheries with conservation and comment on areas that deserve further research.

#### **Seahorses on Danajon Bank**

Upon integrating the evidence from all of my chapters, I conclude that current exploitation by lantern fishers on Danajon Bank threatens seahorse populations and catches. The only record of historic baselines, fisher reports during interviews, suggests that catch rates have sharply declined. *Hippocampus comes* may be more at risk of extinction than its current Vulnerable listing because Red Listing does not consider long-term historic declines (Chapter 2). Based on the available data, it is difficult to be sure of the exact magnitude of these declines (Chapter 3), but corroboration across fishers that declines have occurred and

best estimates of minimum declines (75% from 1970 to 1994, Chapter 2) are cause for concern. That said, lantern fishers are restricted to shallow waters (Chapter 4) and *H. comes* occur well below their reach (Chapter 5), so it is possible that catch rates perceived by fishers declined more quickly than abundances for the entire population (hyperdepletion; Hilborn & Walters, 1992). Even when the total population (shallow and deep) is considered (Chapter 5), however, current exploitation rates (Chapter 4) are still capable of causing alarming declines to a level that warrants threatened status on the Red List. Moreover, the depth refuge is currently at risk of being fished by highly efficient and destructive gears such as trawling and compressor fishing. This quantitative evidence of overexploitation complements qualitative indicators of overfishing that have already been deduced (e.g. juveniles captured; Martin-Smith *et al.*, 2004).

In addition to the management/conservation actions that have already been recommended for this fishery (Martin-Smith *et al.*, 2004), and seahorse fisheries generally (Foster & Vincent, 2005), my work suggests that protecting deep refugia could benefit both fisheries and conservation. No areas deeper than 10 m are protected in Danajon Bank (Hansen *et al.*, in press). Although small refugia have been established in shallow waters, the amount of area they protect (0.32% of DB; Hansen *et al.*, in press) falls short of national law requiring municipalities in the Philippines to protect 15% of their waters (Fisheries Code: RA 8550). The area of protected shallows is also dwarfed by my conservative estimates of depth refuge from lantern fishing area (25% of Danajon Bank, Chapter 5). Narrowing in on exactly which deep areas have seahorses will be very challenging because of the practicalities of diving in such remote locations, at depth, in search of a rare, nocturnal species. Protecting

deep areas identified by fishers as having seahorses or seahorse habitat would be one pragmatic approach.

Prohibiting the most destructive and efficient gears from deeper waters (>10 m) might also help in preventing further collapse of *H. comes* populations. Given the large area in deeper waters and their distance away from shore and the watchful eyes of communities, it may be impractical to enforce or patrol deeper areas to prevent all fishing. Mobile fishing gears (e.g. trawling) that destroy habitat and compressor fishing that has high rates of depletion (Gallardo *et al.*, 1995; Akamine, 2001; Pet & Muljadi, 2001; Morgan & Panes, 2004) are already illegal on most of Danajon Bank. Perhaps the most pragmatic solution, therefore, would be to focus limited enforcement resources on excluding these most destructive and efficient gears from deep refugia.

Monitoring the response of seahorse populations and fisheries to conservation and management interventions will continue to be important. Further research on connectivity among populations would help refine recommendations for the placement and size of refugia (that this far appear not to bolster seahorse populations; Yasue & Vincent, in review) and may become more tractable with newly emerging tagging techniques (reviewed by Jones *et al.*, 2009). Results from tagging studies, combined with the modeling framework I developed (Chapter 5) may even allow predictions about how quickly refugia could help restore populations and fishery catches.

In reality, of course, it is likely to remain more practical to monitor seahorse fisheries than seahorse populations. For example, monitoring spatial fishing behaviour could be an important way to detect the effect of technological improvements on fishing efficiency. The low-tech nature of the lantern fishery leaves it open for increased efficiency known to have

masked increased capacity in other fisheries (e.g. Atlantic cod; Hutchings & Myers, 1994). Since lantern fishers tend to rely on their engines to travel far from home to fish, changes in gas prices could have drastic effects on the amount of area they can access and exploit (Abernethy *et al.*, 2010). Some fishers already report reverting to paddling because they cannot afford gasoline (O'Donnell personal observation).

GPS tracking may be the most effective method for documenting spatial fishing patterns. It does not require fishers to read maps (a challenge for some), it can be ground-truthed easily, and it produces outputs that might re-ignite local interest in conservation. Moreover, GPS tracking technology is rapidly becoming more accurate, less expensive, and more broadly available now that even cell phones (omnipresent in the Philippines) are equipped with GPS tracking capacities.

In general, overfishing on Danajon Bank will only be halted by addressing wider social and governance ills. Over-population, poverty, few income earning options, poor infrastructure and poor governance plague this region and are serious challenges to conservation and food security (Pauly, 2000; Green *et al.*, 2003; Stobutzki *et al.*, 2006; Fernandez, 2009). Fisheries research and management is shifting toward placing emphasis on interactions among people, power, external disturbance and uncertainty, and wider governance dynamics (Andrew *et al.*, 2007). Recent research emphasizes the critical importance of understanding the socioeconomic context of small-scale fisheries for their effective management (Cinner, Sutton & Bond, 2007; McClanahan *et al.*, 2009; Gutierrez, Hilborn & Defeo, 2011). The goal should be to move toward an adaptive management-type system where conservation/management actions are applied, monitored, re-evaluated, and

revised if necessary (Walters, 1986; McClanahan *et al.*, 2009), but all this will need improved governance.

### **Broader implications and research directions**

In addition to shedding light on the status of the lantern fishery and the threatened seahorse *H. comes*, my evaluation of approaches to reconcile fisheries and conservation contributes new insights that suggest future research.

My research helps develop robust approaches to incorporating local knowledge into quantitative assessments. Although the uncertain reliability of local knowledge may be an impediment to its use (White *et al.*, 2005; Brook & McLachlan, 2008), the existence of bias does not render local knowledge useless (Paterson, 2010). In fact, local knowledge should be probed with the same analytical rigor as scientifically collected data, and to do so we need methods to quantify and correct bias in local knowledge. Developing methods of quantifying these biases and/or procedures to correct for them, as I explored in Chapter 2, would allow researchers to interpret interview responses appropriately, improving conservation assessments.

A range of existing techniques should be utilized to help us collect and apply local knowledge as accurately as possible. There is a growing body of literature about how to design studies, phrase questions, and choose appropriate respondents (e.g. Neis *et al.*, 1999a; Davis & Wagner, 2003; Anadón *et al.*, 2009). Quantitative assessment protocols are being developed that explicitly incorporate local knowledge (e.g. Walmsley, Medley & Howard,

2005). It is necessary to acknowledge memory biases that may be present (Daw, 2010) and consider the motivations of respondents to over- or under-report (reviewed by Yasué *et al.*, 2010). Comparing multiple sources of local knowledge with scientific observations can help determine which biases are truly problematic (Chapter 3 and broader methods reviewed by Danielsen *et al.*, 2005). Assumptions about biases should also be made explicit in research papers and researchers should experiment with metrics to account for biases and evaluate effects on results.

The importance of understanding the perceptions of resource users extends beyond filling data gaps in single-species monitoring. Appropriate application of local knowledge requires a broader understanding of community, beliefs, and the economic context of that knowledge (Neis *et al.*, 1999a; Aswani, 2010). For example, research in Chapter 2 is part of a larger project to evaluate options for seahorse conservation and management action. Project Seahorse has consulted with a range of stakeholders and experts from fishers to scientists about preferred management options (Martin-Smith *et al.*, 2004). Understanding the spatial and behavioral dynamics of fishing (Chapter 4) will be part of a movement beyond managing for single species to a more comprehensive social-ecological system (Berkes *et al.*, 2000).

A better understanding of how humans perceive and recall environmental change can help us understand conflicts in conservation and resource governance (Daw, 2010). Conflict can arise when there is a gap between fishers' perceptions and scientific knowledge used to inform management of fishing (e.g. Gray *et al.*, 2008). From another study conducted in the focal community, Handumon, we know that locals perceive their Marine Protected Area (MPA) to be more successful than scientific data suggest (Yasué *et al.*, 2010). Since local

support is a key determinant of ecological success in community-based MPAs, it will be important to acknowledge and then explore this discrepancy in order to ensure long-term viability of the MPA.

A broader incorporation of resource user knowledge into IUCN Red Listing could facilitate rapid assessment of the many exploited and potentially threatened species. Assessing threats to species and evaluating their extinction risk are critical steps toward halting declines. The framework for conservation assessments developed by the IUCN has been thoroughly vetted and is considered the global gold standard (Colyvan *et al.*, 1999; de Grammont & Cuaron, 2006; Rodrigues *et al.*, 2006a). Red List assessments are most commonly based on ecological data (McClenachan *et al.*, in review) and/or scientific ‘expert’ opinion. In developing countries with high biodiversity, heavy reliance on wildlife resources, and rapid environmental change, scientific data collection methods are often too time consuming or expensive (Anadón *et al.*, 2009; Rist *et al.*, 2010) and are therefore impractical (Johannes, 1998; Danielsen *et al.*, 2005). With the number of threatened species on the rise and the vast majority (>95%) of described species yet to be assessed (Vié *et al.*, 2009), traditional scientific methods alone are unlikely to be able to inform species assessments. Supplementing the use of scientifically collected data with fisher knowledge could rapidly increase understanding of the status of marine species.

Retrospective threat classifications, made possible by incorporating local knowledge, highlight a potential limitation to the Red Listing process (Chapter 2). For marine species that have been affected by fishing for at least hundreds of years (Jackson *et al.*, 2001), assessment of contemporary conservation and fishery status must be conducted relative to historic baselines (Pauly, 1995). Because the default time frame for Red List assessments is

only a 10-year window (IUCN, 2008b), there is little capacity for incorporating long-term changes for relatively short-lived species. Therefore, Red Listing may underestimate risk of extinction for historically exploited species (Vincent & Hall, 1996). This shifting baseline in Red Listing should be considered during current efforts to conduct first assessments for marine fishes (e.g. the Global Marine Species Assessment).

The nearly universal (across wide uncertainty in key parameters and processes) benefits of deep reef refugia suggested by my study, emphasize the importance of protecting these areas just out of small-scale fishers' reach. Although we are only beginning to quantify depth refugia on coral reefs, they may actually be widespread (Tyler *et al.*, 2009). If other refugia support population persistence and shallow catches as broadly as in my study, protecting deep (>10 m) reef areas could help safeguard the food supply shallow reefs provide for millions of the world's most poor (Moberg & Folke, 1999). Furthermore, if refugia in other locations are as large as we estimate they could be on Danajon Bank, protecting them could help reach global marine protected area goals (10% by 2020; CBD, 2006). More research is needed to quantify the contribution that un-fished deep reef areas make to the shallows for other species. My work provides pragmatic methods for identifying refugia and evaluating their local importance.

Ultimately, if we are to stem the global decline of fish populations, especially those exploited by small-scale fisheries, we need 'to be creative and innovative and to consider new ideas and approaches' (Berkes *et al.*, 2001). Tapping into unconventional data sources like local knowledge, improving understanding of fisher behaviour, and thinking broadly about how to make precautionary decisions despite uncertainty are pragmatic approaches that



provide key information highly relevant to the futures of coral reef fishes and small-scale fisheries alike.

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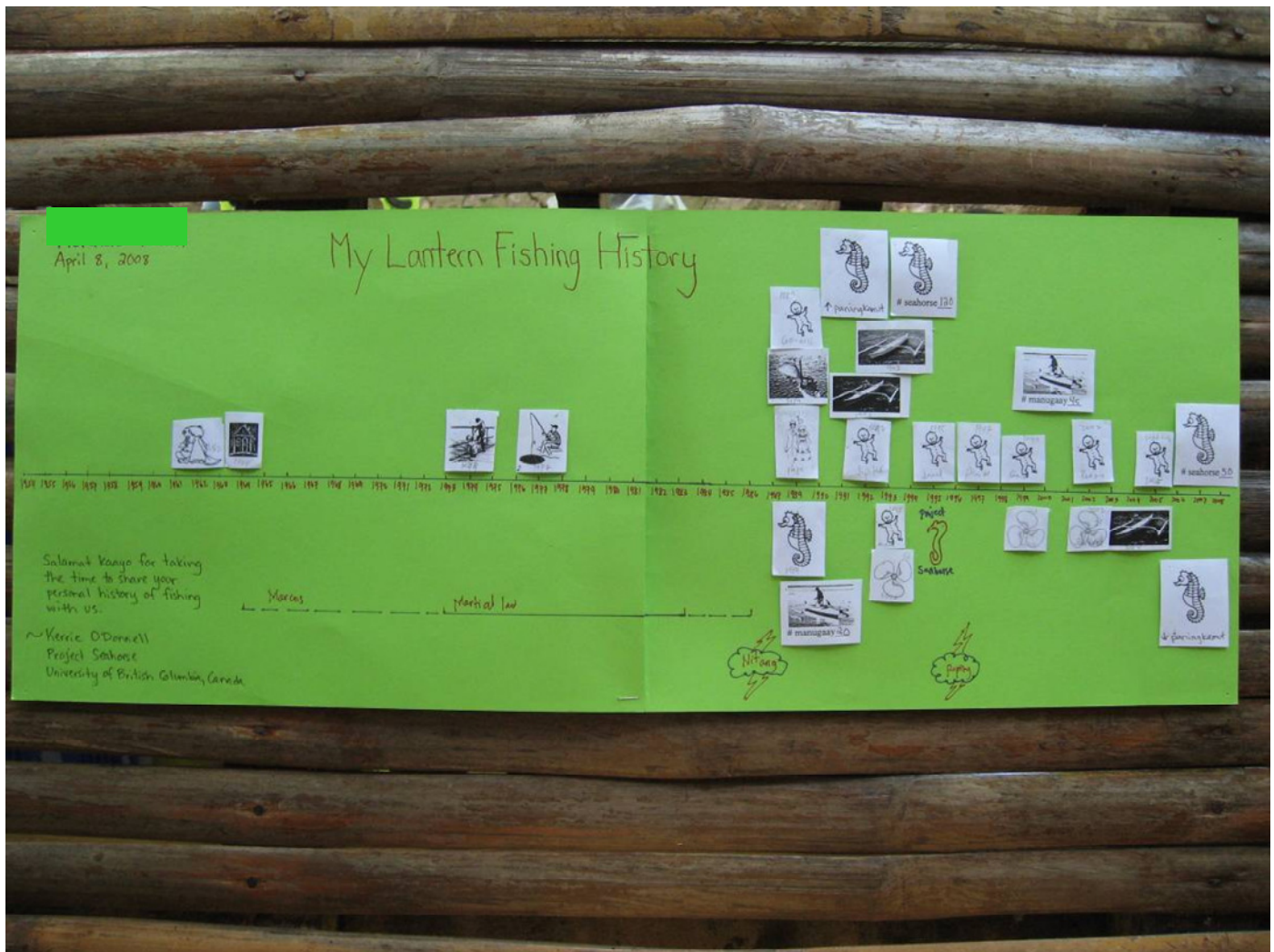
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## **Appendices**

## Appendix A: Example of a fisher's 'personal timeline' visual aide

Figure A.1. Example of a fisher's 'personal timeline' visual aide used during interviews about the history of seahorse catches in Chapter 3. Fisher name blocked out for anonymity purposes. Small pictures representing events in a respondent's life were tacked to the timeline at the start of each interview. In this example the fisher was born just before Marcos became president (~ 1962), began fishing on his own in 1977, was married in 1987, and then in the year he was married he also began lantern fishing, and had his 1<sup>st</sup> child (of 8). The highest number of seahorses he ever caught in one night (120) was in the same year his 4<sup>th</sup> child was born (1995), also the year Project Seahorse began working in his village and also around the same time as the typhoon 'Ruping'. At that time he was putting the most effort (paningkamut) into lantern fishing of any time during his career.



## **Appendix B: Additional information about interviews and comparison between focal communities in Chapter 4**

**Table B.1. Interview schedule for discussions with lantern fishers about diving and fishing limits**

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Biographical information: name, age, number years lantern fishing, number years fishing seahorses.
Q1. What are the names of fishing grounds where you catch seahorses?
Q2. How deep do you dive in each of these places?
Q3. How big is a dupa*?
Q4. What is the deepest you can dive?
Q5. Do you go that deep while lantern fishing?
Q5a. If yes - Do you find seahorses that deep?
Q6. What limits how deep you dive while you are fishing?
Q7. How deep do you think seahorses live and why?
Q8. Do you think there are seahorses down deeper than you fish?

---

\*or 'person height' the local unit of depth measurement. We asked this question to be able to convert to meters.

**Table B.2 We conducted a Wilcoxon rank-sum test (non-normal data) to evaluate whether there were differences in transit, fishing, or dive parameters between our two focal communities of Handumon and Batasan.**

	<i>Transit</i>		<i>Fishing</i>		<i>Dives</i>			
	Total Time	Total Distance Traveled	Total Time	Total Distance Traveled	Max. Dive Depth	Mean Dive Depth	Max. Dive Duration	Mean Dive Duration
Wilcoxon rank-sum test result	(hr)	(km)	(hr)	(km)	(m)	(m)	(s)	(s)
W	15	15	32.5	24	12	2.5	8	6
p	0.74	0.76	<b>0.02</b>	0.33	0.82	0.08	0.65	0.36

## **Appendix C: Evaluation of the swept-area calculation assumption of randomly distributed fishing effort**

### **Data collection and analysis**

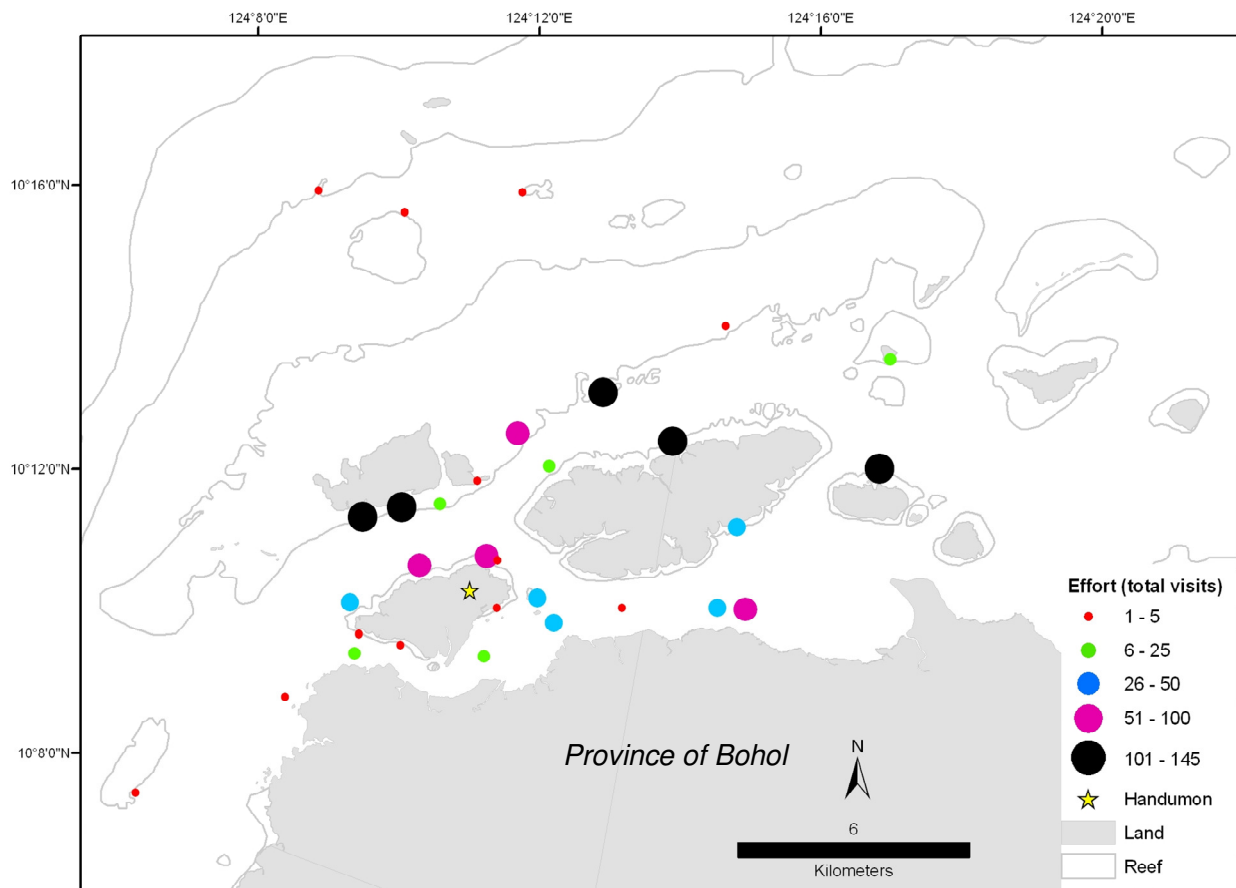
To evaluate the assumption that fishing effort was randomly distributed in space, we turned to the only data available - fisher logbooks completed for an entire year in Handumon. Logbooks were distributed by Filipino Project Seahorse team members who followed up with discussions and log collection at the end of each month (Vincent *et al.*, 2007). Here, we focused on fishers' record of the names of fishing grounds they visited. This program ran for many years in Handumon (though not in Batasan), but we chose to analyze data from 2001 because it was the only year when data were recorded in all months. Logbooks were completed by 12 fishers in 2001, representing about 40% of the lantern fishers in Handumon at that time.

To place fishing ground names on a map we used two approaches. First, in November 2004 most (27) fishing grounds were GPS located with the aide of an experienced Handumon lantern fisher. If a fishing ground named in fisher logbooks had not been GPS located and was the name of a community or island (4 fishing grounds), we placed the fishing ground location just offshore from the center of that community. We were able to locate a total of 31 of the 50 fishing grounds mentioned in 2001 logbooks accounting for 89% of the total nights fished. We calculated straight-line distance to each fishing ground from a central point in Handumon using an ArcView script (ESRI, 2010) trends. We also calculated the coefficient of dispersion (variance / mean, Begon, Harper & Townsend, 1996) between fishing grounds to categorize the type of distribution.

## Results

Lantern fishing effort from Handumon in 2001 was not randomly distributed, but aggregated at preferred fishing grounds. Handumon lantern fishers recorded 1,103 nights of fishing during 2001 on grounds that extended up to 12.5 km from Handumon, with the highest concentration of effort seen 3 to 6 km from shore (Fig. S1). The majority of effort was confined to a small proportion of the 31 fishing grounds visited. The five (16%) most visited fishing grounds (Fig. S1, largest black circles) accounted for more than 50% of the total nights fished, nearly 75% to total fishing effort applied. There were on average 34.5 visits per fishing ground with a dispersion coefficient of 55.9, which was significantly higher than 1, therefore, the overall effort allocation aggregated in space.

**Figure C.1. Distribution of fishing effort (total nights) at each fishing ground visited in 2001 from Handumon lantern fisher logbooks.**





## **Appendix D: R code to run depth refuge model and supplementary analyses**

Below are all the equations used in the model presented in Chapter 5, pseudocode, and the R code for running the model over the range of parameter combinations I evaluated. In addition, I present figures that provide evidence suggesting conclusions drawn from the model, even after running it for just ten years, are robust and I discuss possible explanations for the refuge causing lower total (shallow + deep) catches. More detailed definitions of indices and parameters are available in Chapter 5, Table 2. For ease of explanation, equations were presented in a slightly different order and numbered differently in the text than the equations below. The equation numbers in the R code, refer to these equation numbers, not those in the text of Chapter 5.

### **Equations:**

$$(1) S_a = e^{-(a-1)M}$$

$$(2) L_a = \begin{cases} L_\infty [1 - e^{-K(a-a_0)}]; & a > a_0 \\ 0; & a \leq 0 \end{cases}$$

$$(3) V_a = \begin{cases} \frac{1}{1 + e^{-a}}; & L_a \geq 70 \\ 0; & L_a < 70 \end{cases}$$

$$(4) f_a = \begin{cases} \frac{b(-2.69 + 3.14 \times L_a)}{2}; & L_a \geq 93 \\ 0; & L_a < 93 \end{cases}$$

$$(5) \phi = \sum_{a=1}^{35} f_a S_a$$

$$(6) E'_i = \phi R'_i$$

Initialize for month 1 for  $a = 1, \dots, 35$  and  $i = 1, 2$  and continue with no fishing until  $N_t$  reaches equilibrium:

$$(7) N_{1ai} = R'_i S_a$$

Repeat for  $t = 1, \dots, n$ :

$$(8) N_{ti} = \sum_{a=1}^{35} N_{tai}$$

$$(9) N_t = \sum_{i=1}^2 N_{ti}$$

$$(10) E_t = \sum_{a=1}^{35} \sum_{i=1}^2 f_a N_{tai}$$

$$(11) C_{ti} = U_i \sum_{a=1}^{35} V_a N_{tai}$$

$$(12) C_t = \sum_{i=1}^2 C_{ti}$$

$$(13) E_{ti} = p_i E_t; \quad i = 1, 2$$

If  $t < n$ , for  $a = 1, \dots, 35$  and  $i = 1, 2$ :

$$(14) N_{t+1,ai} = \begin{cases} \frac{\rho(R'_i / E'_i) E_{ti}}{1 + [(\rho - 1) / E'_i] E_{ti}}; & a = 1 \\ N_{t,a-1,i} e^{-M} (1 - V_{a-1} U_i); & a = 2, \dots, 35 \end{cases}$$

Assign equilibrium no fishing  $N_{tai}$  to be the first month's fish in fished simulation which will repeat equations (8) through (14) for ten years (120 months).

## R Code:

```
##### R CODE: Set up depth refuge model to run across many different parameter
##### combinations. First dim parameters, then let model run
##### to equilibrium without fishing, then start fishing and let model run for
##### 10 years with fishing.

#####LIST OF PARAMETERS TO CALCULATE ACROSS #####

param.options <- expand.grid(a0 = 0.36, b = 1.79, K = 0.24, Linf = 167, M =
c(0.06, 0.1, 0.15),
n = 120, p = seq(0,1,0.1), Rp1 = c(49063, 98162, 147189),Rp2 = c(147189,
98162, 49063),
U1= c(0,0.03,0.04,0.05,0.06), U2= c(0,0.03,0.04,0.05,0.06), rho =
c(1.1,1.5,2,5,7),.n=5000)

##### SET UP MODEL TO RUN SIMULATIONS #####

runSim2 <- function(param.options) {      # dim or set up all variables

  ##dim params in param.options
  a0 <- as.numeric(param.options[1])
    # as.numeric () a0 is a single value which describes age at size 0
  b <- as.numeric(param.options[2])
    # b single value, slope in VB growth curve
  K <- as.numeric(param.options[3])
    # K single value, VB growth parameter
  Linf <- as.numeric(param.options[4])
    # Linf, single value, max size
  M <- as.numeric(param.options[5])
    # as.numeric() M mortality rate, single value
  n <- as.integer(param.options[6])
    # as.integer() n, number of time steps, single value
  p <- c(as.numeric(param.options[7]), 1-as.numeric(param.options[7]))
    # p, proportion of recruits going to shallow
  Rp <- as.numeric(param.options[8:9])
    # Rp, vector of 2 values, number of recruits to shallow/deep at
    #equilibrium
  U <- as.numeric(param.options[10:11])
    # U, exploitation rate (U1) shallow, (U2) deep
  rho <- as.numeric(param.options[12])
    # rho, recruitment compensation parameter, single value
  .n <- as.numeric(param.options[13])
    #n for no fishing equilib simulations

  ##set up params and seed model with starting population
  tt <- 1:n
    #time index and params not already 'dimmed'
  .tt <- 1:.n
    #for no fishing equilib simulations
  N <- array(data = NA, dim = c(n,35,2))
  .N <- array(data = NA, dim = c(.n,35,2))
    #for no fishing equilib simulations
  Nt <- rep(0,n)
  .Nt <- rep(0,.n)
    #for no fishing equilib simulations
  Eti <- array(data = NA, dim = c(n,2))
  .Eti <- array(data=NA, dim =c(.n,2))
    #for no fishing equilib simulations
  Cti <- Eti
  .Cti <- .Eti
    #for no fishing equilib simulations
```

```

Ct <- Nt; Et <- Nt; Nt1 <- Nt; Nt2 <- Nt; Ct11 <- Nt; Ct12 <- Nt
.Ct <- .Nt; .Et <- .Nt; .Nt1 <- .Nt; .Nt2 <- .Nt; .Ct11 <- Nt; .Ct12 <- Nt

a <- 1:35
Sa <- exp(-(a-1)*M)
      # (Eq. 1) these equations numbers refer to equations in the #appendix
not equations #s in the text of Ch 5.
La <- Linf*(1-exp(-K*(a-a0))); La[La<0] <- 0 # (Eq. 2)
Va <- 1/(1+exp(-a)); Va[La<70] <- 0 # (Eq. 3)
fa <- floor(b*(-2.69+3.14*La)/2); fa[La<93] <- 0 # (Eq. 4)
phi <- (sum(fa*Sa)) # (Eq. 5)
Ep <- phi *Rp # (Eq. 6)

.N[1,,1] <- Sa * Rp[1] # (Eq. 7)
.N[1,,2] <- Sa * Rp[2] # (Eq. 7)

##### INITIALIZE MODEL BY PREDICTING POPULATIONS SIZE UNTIL COMES TO EQUILIBRIUM
WITHOUT FISHING #####

equil.value <- 11
.tt = 1
Continue = TRUE
while(Continue){ # tt = 12
  .Nt1[.tt] <- sum(.N[.tt,,1], na.rm = TRUE) # (Eq. 8)
  .Nt2[.tt] <- sum(.N[.tt,,2], na.rm = TRUE) # (Eq. 8)
  .Nt[.tt] <- .Nt1[.tt] + .Nt2[.tt] # (Eq. 9)
  .Et[.tt] <- round(sum(fa * .N[.tt,,], na.rm = TRUE)) # (Eq. 10)
  .Ct11[.tt] <- sum(.N[.tt,,1] * 0 * Va, na.rm = TRUE) # (Eq. 11)
  .Ct12[.tt] <- sum(.N[.tt,,2] * 0 * Va, na.rm = TRUE) # (Eq. 11)
  .Ct[.tt] <- .Ct11[.tt] + .Ct12[.tt] # (Eq. 12)

  for (i in 1:2){
    .Eti[.tt,i] <- p[i]*.Et[.tt] # (Eq. 13)
  }

  if (.tt<.n){
    for (i in 1:2){
      .N[.tt+1,1,i] <-
        (rho*(Rp[i]/Ep[i])*Eti[.tt,i])/(1+((rho-
          1)/Ep[i])*Eti[.tt,i]) # (Eq. 14)
      Nt.holder <- .N[.tt,-length(a),i]*exp(-M)*(1-Va[
        length(a)]*0) # (Eq. 14)
      .N[.tt+1,-1,i] <- Nt.holder
    }
  }
  if(.tt> equil.value){
    if(all(.Nt[.tt]==.Nt[(.tt-equil.value):(.tt-1)]) | .tt >=
      5000) Continue = FALSE
  }
  .tt <- .tt + 1
}

N[1,,1] <- .N[.tt,,1]
N[1,,2] <- .N[.tt,,2]
#Assign equilib. No fishing numbers at age and zone to be the first month's
fish in fished simulation below

##### START FISHING AND PREDICT POPULATIONS FOR 10 YEARS #####

for (tt in 1:n){
  Nt1[tt] <- sum(N[tt,,1], na.rm = TRUE) # (Eq. 8)
  Nt2[tt] <- sum(N[tt,,2], na.rm = TRUE) # (Eq. 8)

```

```

Nt[tt] <- Nt1[tt] + Nt2[tt] # (Eq. 9)
Et[tt] <- round(sum(fa * N[tt,,], na.rm = TRUE)) # (Eq. 10)
Ctil[tt] <- sum(N[tt,,1] * U[1] * Va, na.rm = TRUE) # (Eq. 11)
Cti2[tt] <- sum(N[tt,,2] * U[2] * Va, na.rm = TRUE) # (Eq. 11)
Ct[tt] <- Ctil[tt] + Cti2[tt] # (Eq. 12)

for (i in 1:2){
  Eti[tt,i] <- p[i]*Et[tt] # (Eq. 13)
}

if (tt<n){
  for (i in 1:2){ # i = 1

    N[tt+1,1,i] <- (rho*(Rp[i]/Ep[i])*Eti[tt,i])/(1+((rho-
      1)/Ep[i])*Eti[tt,i]) # (Eq. 14)
    Nt.holder <- N[tt,-length(a),i]*exp(-M)*(1-Va[
      length(a)]*U[i]) # (Eq. 14)
    N[tt+1,-1,i] <- Nt.holder
  }
}

vals.2.return <- c(M = M, p = p, rho = rho, Rp = Rp, U = U)
#return value of params in that model run
data.2.return <- c(s.Nt120 = Nt[length(Nt)]/Nt[1], s.Nt120.1 =
Nt1[120]/Nt1[1], s.Nt120.2 = Nt2[120]/Nt2[1], cum.Ct120 = sum(Ct),
cum.Ct120.1 = sum(Ctil), cum.Ct120.2 = sum(Cti2))
#return proportion of pop remaining and cumulative catches
return.these1 <- c(vals.2.return, data.2.return)
return.these2 <- cbind(Nt = Nt, Nt1 = Nt1, Nt2 = Nt2, Ct = Ct, Ctil = Ctil,
Cti2 = Cti2)
#return time series too
return(list(return.these1, return.these2))

} # End of runSim2

##### RUN THE MODEL OVER RANGE OF PARAM VALUES #####

model.results <- list()
for(i in 1:nrow(param.options)){
  model.results <- c(model.results, runSim2(param.options[i,]))
}

##### CALL RESULTS OF THE MODEL AND REFORMAT (ex. Only for param.results so can use
it for plotting #####

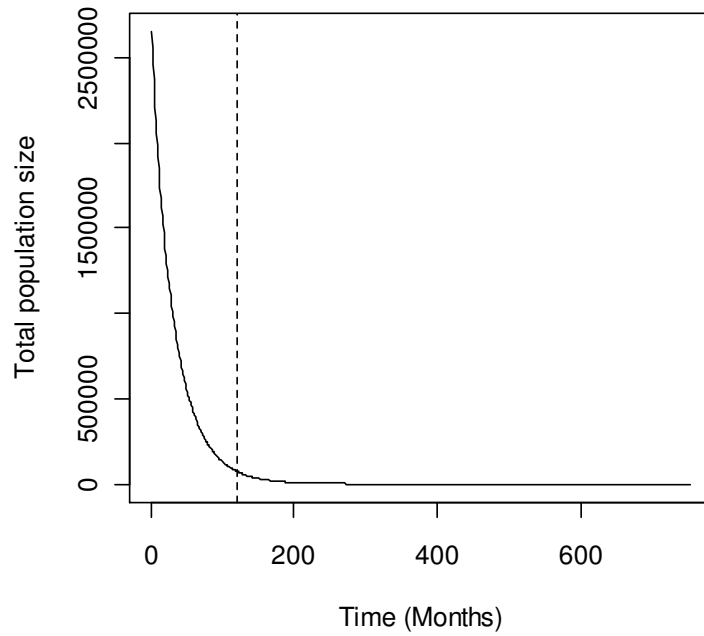
param.results <- model.results[seq(1, length(model.results), 2)]
#pick all the odd ones out of model.results - these are params and
data.2.return

rem.names <- names(param.results[[1]]) #Remember names
param.results.format <- unlist(param.results)
names(param.results.format) <- NULL #ixnay the names
dim(param.results.format) <- c(length(rem.names), length(param.results))
#get all the same values into the same ROWS
param.results.format <- t(param.results.format)
#Transpose the table so that the same values are now in Columns -
#more intuitive
colnames(param.results.format) <- rem.names #name the columns
param.results.format <- as.data.frame(param.results.format)
#Convert to data.frame
typeof(param.results.format) #check, should say list
head(param.results.format) #check

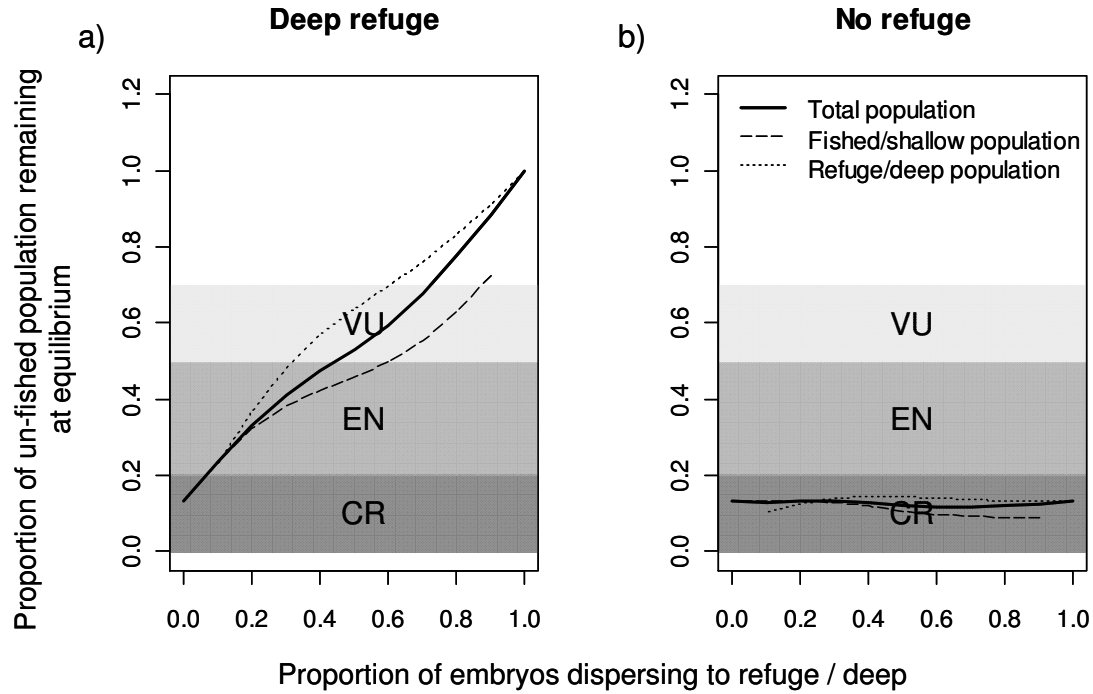
```

## Supplementary Figures

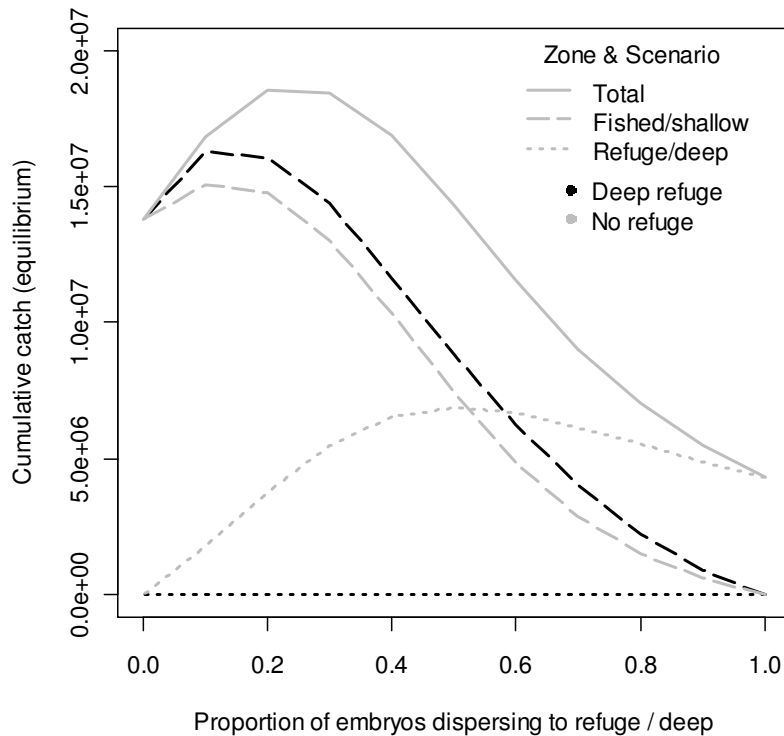
**Figure D.1.** Trajectory of a population at the combination of parameters that caused the most severe decline for comparison of amount of decline at 10 years (120 months, dashed line) and at equilibrium (758 months).



**Figure D.2.** Proportion of total (solid line), shallow (dashed line), and deep (dotted line) *H. comes* populations remaining at equilibrium at default parameter values and across a range of uncertainty about the proportion of embryos that disperse to the deep. We compare two scenarios: 1) a small, deep refuge where 25% of the un-fished equilibrium population was assigned to the refuge and 75% were fished at current exploitation rate (a), and 2) no refuge where 100% of the un-fished equilibrium population was fished at current exploitation rate (b).



**Figure D.3.** Comparison of cumulative catches of *H. comes* at equilibrium from the shallow (dashed), deep (dotted), or total (shallow+deep; solid line) zones. We compare two scenarios: 1) a deep refuge (black lines) and 2) no refuge (grey lines).



## Effect of the refuge on total catches

This study was not designed to evaluate whether establishing a reserve can perform as well as other fishery management tools and increase fishery yield despite a loss of fishing area to the refuge. That said, it is interesting that in contrast to other models of MPA effectiveness that were designed for this purpose, we found that the presence of a refuge decreased total fishery catches (Fig. 5.5), even when the model was run until fishing came to equilibrium with the population (Fig. D.3). Other models similar in structure to ours suggest that when an MPA is established total catches can remain the same (Hastings & Botsford, 1999) or even increase (White & Kendall, 2007), despite fishing area being diminished. The life history of *H. comes* may have contributed to this difference. For example, it has been suggested that increases in yield using reserves are less likely for short-lived as compared to long-lived species (Gaylord *et al.*, 2005).