RARE SEAHORSES HAVE BIG IMPLICATIONS FOR SMALL FISHES IN BYCATCH

by

Julia Margaret Lawson

B.Sc., Dalhousie University, 2010

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF

THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

in

THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

(Zoology)

THE UNIVERSITY OF BRITISH COLUMBIA

(Vancouver)

August 2014

Abstract

The incidental capture of marine organisms is a critical area of concern in fisheries management, and despite the dominance of small fishes in bycatch biomass around the globe, we know little about the impacts of bycatch on these small fishes. Here, I focus on one such small fish taxon, the seahorses (*Hippocampus* spp.). Seahorse populations are typically patchy, and they have a specialized life history that makes them more likely to be vulnerable to overfishing. I assessed the impacts of nonselective gear on seahorses in one country by examining potential impacts for three species in peninsular Malaysia. I also assessed the impacts of seahorse bycatch globally, by synthesizing a global array of studies. For my national analysis, I used data-poor assessment methods to estimate life history parameters for three species of seahorse in peninsular Malaysia (Chapter 2). For my global analysis, I extracted data found in a published and unpublished reports to generate a synthesis of global seahorse bycatch (Chapter 3). Nationally, my findings indicate the potential for overfishing for one species found in the southwest region of the peninsular Malaysia. Globally, my synthesis revealed that all fishing gears obtained seahorses in bycatch, but at very low rates of capture per vessel per day. Across all countries, fishers characterized seahorses as relatively rare in their bycatch and declining in catches. One key finding was that low CPUE scaled up to tens of millions of seahorses obtained globally as bycatch each year. My results address the impacts of bycatch on seahorses and discuss the implications for other small bycaught fish species, especially those that are demersal or rare.

Preface

My co-supervisors, Drs. Amanda Vincent and Sarah Foster, and myself developed the research questions and methodological design of this thesis collaboratively. I collected all of the data used for Chapter 2 of this thesis with the help from my research assistant. Other researchers from Project Seahorse and elsewhere collected the data used in Chapter 3. I carried out all analyses and prepared all manuscripts in this thesis.

A version of Chapter 2 has been accepted for publication. I am the lead author, along with my co-supervisors Drs. Amanda Vincent and Sarah Foster, and my collaborators at the University of Malaya, Dr. Chong Ving Ching and Mr. Adam Lim Chee Ooi. I conducted all of the research for Chapter 2, and wrote the paper in collaboration predominantly with Drs. Sarah Foster and Amanda Vincent. Dr. Chong Ving Ching and Mr. Adam Lim Chee Ooi provided essential logistical, scientific and cultural guidance during field data collection, which was essential to the success of this chapter. My research assistant, Leong Yun Sing, also played an invaluable role in data collection for Chapter 1, as she translated and provided essential cultural support.

A version of Chapter 3 is currently in review for publication. I am the lead author, along with my co-supervisors Drs. Amanda Vincent and Sarah Foster. I conducted the analysis for Chapter 3 and wrote the paper in collaboration with Drs. Sarah Foster and Amanda Vincent. The data for Chapter 3 was collected during a series of fisheries and trade surveys led by Project Seahorse researchers – Julia Baum, Brian Giles, Allison Perry, Marivic Pajaro, Andres Cisneros-Montemayor, Ting-Chun Kuo, Jessica Meeuwig and Jana McPherson. Published and unpublished reports from researchers not affiliated with Project Seahorse were also used in the analysis, as described in Chapter 3.

The field research conducted in Chapter 2 was approved by the UBC Animal Care Committee, Permit A12-0288, project title: “Creating momentum for seahorse populations in Southeast Asia” and UBC Behavioural Research Ethics Board, Permit H12-02731, project title: “Fishing and trade in Southeast Asia”. Chapter 3 was a synthesis of many studies, and the details on ethics clearance can be found in the original papers.

Table of Contents

List of Tables

List of Figures

Acknowledgements

Foremost I would like to acknowledge how lucky I was to be guided and supported throughout this thesis by two strong and inspiring women, my supervisors Drs. Amanda Vincent and Sarah Foster. Thank you for having faith in my abilities and giving me the opportunity to undertake an incredible journey in Southeast Asia (and as equally an incredibly journey in the office in Vancouver!). You both introduced me to the world of marine conservation and pushed me towards creating tangible outcomes from this research. It’s been a rewarding experience. Thank you as well to Drs. Peter Arcese and Eric Taylor who, as members of my supervisory committee, provided useful advice and support to me throughout the course of this research.

During my time in peninsular Malaysia, I was lucky to receive invaluable support from Adam Lim Chee Ooi and Dr. Chong Ving Ching who made me feel welcome and supported in a completely new culture. They provided essential logistical, cultural and scientific support that was critical to the success of my second chapter. The success of my second chapter was also hugely reliant on my incredible research assistant, Leong Yun Sing, who worked with me throughout the four months that I was in Malaysia. Her roles were numerous – translator, driver, tour guide, dive buddy and colleague to name a few – her ability to charm fishers into helping us with our project was unparalleled. Lastly, I would like to thank the late Choo Chee Kuang who laid the foundations of seahorse research in Malaysia before his untimely death in 2013.

This thesis is a contribution from Project Seahorse. Chapter 2 was made possible thanks to collaborations with the Institute of Earth and Ocean Science at the University of Malaya, and was supported by grants from The Explorers Club and the University of British Columbia’s Biodiversity Research Integrative Training & Education (BRITE) program. Support provided by the John G. Shedd Aquarium in Chicago, Guylian Chocolates Belgium and an anonymous donor was also critical in advancing the goals of this thesis. Chapter 3 was made possible thanks to efforts by many scientists, some who worked as part of Project Seahorse, and other researchers from elsewhere. In particular, I would like to thank Joanna Alfao-Shigueto and Jeffrey Mangel from ProDelphinus for sharing unpublished data from Peru that was supported by funds from Duke University and the Oak Foundation.

I would not be where I am today without the support of my incredible Fisheries Centre friends – thank you for keeping me sane and putting up with me through it all. The most important thank you goes to Lauren Weatherdon, my 'kindred spirit' to whom I was attached at the hip from day one of FISH 520. Thank you for being my rock through this journey, I'm so excited that we've made it through together – ¡Salud! I was also incredibly lucky to have enduring support from Project Seahorse staff and students (current and former) – Lindsay Aylesworth, Stefan Wiswedel, Kyle Gillespie, Ting-Chun Kuo, Danika Kleiber, Jenny Selgrath, Tse-Lynn Loh, Ally Stocks, Xiong Zhang, Tanvi Vaidyanathan, Riley Pollom, Scott Finestone, Gina Bestbier and Tyler Stiem. A huge thank you in particular to Lindsay Aylesworth for talking seahorses any time, any place, whether it was Koh Tao, Thailand or Kuala Lumpur, Malaysia. Additional thanks to several other Fisheries Centre colleagues – especially Andres Cisneros-Montemayor, Catarina Wor and Tom Carruthers for your support at various points throughout my thesis.

Finally, I'd like to thank my family – Keith, Christy, Stefan, Anna and Sam, for your endless encouragement, both emotionally and financially, throughout all of my travels, explorations and projects in the marine world so far. Thank you for allowing me to take this path.

To my parents, Keith and Christy

When you are inspired by some great purpose, some extraordinary project, all your thoughts break their bonds: Your mind transcends limitations, your consciousness expands in every direction, and you find yourself in a new, great and wonderful world. Dormant forces, faculties and talents become alive, and you discover yourself to be a greater person by far than you ever dreamed yourself to be. – Patañjali

Chapter 1: Introduction

1.1 Rationale

The goal of this thesis was to assess local and global impacts of bycatch on small fishes, using seahorses (*Hippocampus* Rafinesque 1810 spp.) as a case study.

1.2 Background

Although the current state and trajectory of the world's fisheries remains under heated debate (Jackson *et al.*, 2001; Worm *et al.*, 2006; Worm *et al.*, 2009; Branch *et al.*, 2011), it appears that global catches have declined despite a global expansion of fisheries (Swartz *et al.*, 2010) and increased fishing effort (Watson *et al.*, 2013). Modern fishing gears rarely catch only target species, and the wasteful practices of many non-selective fishing gears present a major challenge to the long-term sustainability of these fisheries. Much of the catch is often composed of non-target species (bycatch). Some of this bycatch may be discarded at sea or at the dock but some or all of it may be retained, for direct sale or for reduction into other products. Studies have estimated that 7.3 million tonnes of bycatch is discarded annually (Alverson, 1994; Kelleher, 2005) but the portion that is retained is unknown. Bycatch is widespread among gear types and species, ranging from pelagic sharks on hooks to marine invertebrates in fish traps.

Bycatch may affect the health of marine fish populations through direct or indirect morality, and can affect ecosystem functioning through community disturbance and habitat damage (Andrew & Pepperell, 1992; Kumar & Deepthi, 2006). Fishing mortality can occur directly, through removal or death of individuals, or indirectly if mortality occurs after discarding. Discarded animals may suffer physiological or physical injuries or be consumed by scavengers like seabirds, dolphins and sharks (Hill & Wassenberg, 2000; Wilson *et al.*, 2014).

They may also experience disruption to their growth, reproductive cycles, or mating and social systems (Rowe & Hutchings, 2003). Even if the non-target species manage to escape capture unharmed, destructive gear can cause considerable long-term damage to marine habitats (Freese *et al*., 1999; Watling, 2014), such as reducing structural complexity, thereby increasing chances of predation (Jennings & Kaiser, 1998). By disturbing bottom habitat these gears can also restructure ecosystems, causing smaller species with higher resilience to become more dominant (Duplisea *et al*., 2002).

Research on the impacts of fishing on bycaught species has been biased towards developed countries and towards charismatic megafauna (Hall, 1997). Research that has been directed at understanding the impacts of nonselective fishing practices has occurred in developed nations, such as northern Australia (Stobutzki *et al*., 2001a; 2001b), the North Sea, North Atlantic, and the United States Gulf of Mexico (reviewed in Alverson, 1994) due to a greater capacity for management and research. However some of the most indiscriminate and destructive fishing gears, often targeting species in shallow waters using small mesh sizes (Kelleher, 2005), are concentrated in developing nations with high but rapidly declining fish biomass, low control on exploitation rates and poor management capacity (Worm & Branch, 2012). Additionally, a significant amount of research on bycatch has been very limited taxonomically. Most has focused on bycatch of charismatic megafauna, such as marine mammals, sea turtles, or sharks (as reviewed by Lewison *et al*., 2004). Some has focused on high-value megafauna such as white marlin (*Kajikia albida*: Kitchell *et al*., 2004), or juvenile yellowfin and bigeye tuna (*Thunnus albacares* and *T. obesus*: Bailey *et al*., 2013). Virtually none has focused on small species of limited value. A shift in research priorities towards developing nations and towards the species most frequently obtained in bycatch is needed.

Bottom trawl fisheries are the least selective gear types. Worst among them are bottom trawls targeting penaeid shrimp, where the majority of the catch biomass is composed of non-target small fish species, greatly exceeding the biomass of targeted shrimp (Andrew & Pepperall, 1992; Kelleher, 2005). The bulk of the biomass obtained in shrimp trawls is composed of small fishes maturing at less than 20 cm and weighing less than 100 grams (Alverson, 1994; Kelleher, 2005). The last global assessment of bycatch discards reported that most of these small so-called 'trash' fishes were discarded, and that discards from shrimp trawls were the highest across all gear types (Kelleher, 2005). While these small fishes were historically treated as low value catch, they are increasingly being retained for reduction into feed for agriculture and aquaculture (Funge-Smith *et al*., 2005; Kelleher, 2005). This retention means that previously high discard rates have dropped nearly to zero in some areas. While this shift reduces the wasteful practices of bycatch fisheries (Andrew & Pepperell, 2002), increased utilization of 'trash fish' pushes bottom trawl fisheries that originally targeted shrimp beyond economic extinction (Lobo *et al*., 2010) allowing these fisheries to continue operation despite the collapse of their target species.

The ecosystem impacts caused from the extraction of large quantities of small fishes have been poorly studied (Chuenpagdee *et al*., 2006), which is concerning given the crucial role small fishes play in food webs and for human food security (Hall *et al*., 2000). Small fishes are a vital link in the marine food chain that supports upper-level carnivores (Alder *et al*., 2008) and their depletion, at least through target fisheries, has been linked to a decline in upper trophic levels (Goñi, 1998). Small fishes act as a vital protein source in developing nations (Hall & Mainprize, 2005), where they can be obtained through low-tech gears and are easy to preserve (Alder *et al*., 2008). Despite their importance, little is known about the impacts of tropical shrimp trawls on the millions of small fishes they catch each year.

1.2.1 Approaches to data-poor fisheries

Previous research on small fishes in bycatch has focused on quantifying bycatch and documenting species, but few studies have attempted to estimate abundance and productivity of these stocks (Foster & Vincent, 2010b). Information on discarded biomass, ratios of bycatch to target fisheries and other raw numbers have raised concern about the unmonitored and large-scale removal of small fishes in bycatch (Kelleher, 2005), but this information has provided little guidance to mangers or scientists as to the status of these stocks. Traditional stock assessment methods used to estimate abundance and productivity fall short for small fishes in bycatch because of a dearth of long-term data. Other approaches are needed to assess the response of small fish populations to bycatch.

Even with few data, information on species' life history, ecology and population parameters can be used to deduce the vulnerability of bycaught organisms to fishing impacts (Dulvy *et al*., 2004, Costello *et al*., 2012). Life history rates of marine fishes vary depending on how an individual organism allocates its energy towards growth, reproduction, survival and movement, which is closely linked to the rate of population increase (Hutchings, 2003). Comparative life history studies have shown that under equal fishing pressure, large, slow-growing, and late maturing species are intrinsically more vulnerable to decline and slower to recover, compared to small, fast-growing, and earlier maturing species (Jennings *et al*., 1998; Jennings, 2005). While such analyses have provided a good rule of thumb, we urgently need to understand the actual susceptibility of small fishes that are demersal, rare, sedentary or social to fishing pressure (Dulvy *et al*., 2003, Stobutzki *et al*., 2001b).

Key life history parameters, such as maximum length or length at maturity, have grounded several data-poor assessment methods. Froese (2004) created three indicators - all based on length at maturity - to assess a catch for status and trends that may signal overfishing in a given population. Dulvy and Reynolds (2002) used maximum body size to effectively predict extinction risk for bycaught skates and rays (Batoidea). Stobutzki *et al*. (2001b) combined life history information with ecological criteria, like depth and habitat preferences, to prioritize shrimp bycatch for research and management. The approach developed by Stobutzki *et al*. (2001b) may be particularly useful for multi-species assessments like bottom trawl bycatch, while rule-of-thumb life history approaches work most effectively when assessing related species.

Information from trade surveys (and associated port surveys) can fill critical gaps in our understanding of threats to bycaught species. Data intensive population assessments or catch-per-unit-effort estimates are likely impossible to generate for the majority of small, bycaught fishes (Foster & Vincent, 2010b), and there are often few formal records on landings. In response, trade surveys and analyses have generated crucial information of considerable reliability and utility. In trade and port surveys, data are often collected from fisher interviews. Such information can be used to crudely estimate catch-per-unit effort (i.e. Baum *et al*., 2003; Meeuwig *et al*., 2006) and can be combined with official fisheries data on fleet sizes to estimate total volumes extracted in bycatch (i.e. Baum & Vincent 2005; Giles *et al*., 2006). Data inferred from fisher interviews must be interpreted with caution, as O'Donnell *et al*. (2010) showed that fishers tend to exaggerate fishing rates and report greater variability in catches during interviews. Nonetheless, when it comes to assessing small fishes in bycatch, imperfect data is better than no data at all (Johannes, 1998).

Concerns about a lack of data extend to a huge assortment of small fish taxa obtained as bycatch around the globe. Information is severely limited even for the small fish species that contribute the greatest biomass to global bycatch discards: (*Leiognathidae* (ponyfish), *Nemipteridae* (threadfin), *Trichurius* sp. (hairtails), *Decapterus* sp., *Saurida* sp.) as noted by Kelleher (2005). We lack even basic life history parameters – including maximum size and size at maturity – for many species within these five taxa (Froese & Pauly, 2014). Seahorses are one taxon of small bycaught fish where a concerted effort has generated information on seahorses in bycatch and seahorse life history.

Information obtained from data-poor fisheries can draw attention to the impacts of destructive and indiscriminate fishing practices on small bycaught fishes – either narrowly, by identifying bycatch impacts on a particular species (Chapter 2), or broadly by producing a global perspective on a particular small fish taxon (Chapter 3). My thesis focused on gathering such information for seahorses (*Hippocampus* spp.), small bycaught fish that can be used as an important flagship species for other small fishes in bycatch.

1.2.2 Seahorses in bycatch

As a model of data-poor small fishes that are caught in bycatch, I focused on seahorses (*Hippocampus* spp.), a genus of 48 species of small fish. They are considered data-poor as currently 75% of seahorse species are listed as 'Data Deficient' on the International Union for Conservation of Nature (IUCN) Red List, meaning that limited data has prevented a reliable assessment of extinction risk for these species (IUCN, 2013). The majority of those that have been assessed are listed as Vulnerable, because of inferred population decline resulting from incidental capture, targeted catch and habitat loss.

Threats to seahorses have been recognized and researched over the past decade, in part because seahorses are charismatic species of significant cultural and medicinal value. The unique appearance of seahorses has inspired works of art for hundreds of years (Scales, 2009) and has created demand for seahorses as curiosity items, souvenirs and for aquarium display. Seahorses are also important components of traditional medicines in many cultures, particularly in traditional Chinese medicine and in Indonesia's *jamu* medicine. In traditional Chinese medicine, seahorses are used powdered or dried to treat a wide range of ailments from asthma to impotence (Vincent, 1996). The combination of these consumer demands has fueled a large global trade in seahorses.

The international trade in seahorses is widespread and involves both dried and live seahorses (Vincent *et al*., 2011b). This global trade was first described during field surveys in Asia conducted between 1993 and 1995 (Vincent, 1996). These surveys uncovered the large-scale international trade in seahorses and drove the creation of Project Seahorse in 1996 (www.projectseahorse.org). Following these original surveys, a series of trade surveys were conducted between 1998 and 2001 in countries within and outside of Asia, uncovering an even larger trade network (McPherson & Vincent, 2004; Baum & Vincent 2005; Martin-Smith & Vincent 2006; Giles *et al*., 2006; Perry *et al.*, 2010; Vincent *et al*., 2011a). Most of the live trade is sourced from Asia but sent to the European Union or North America, while Asia remains the leading importer and exporter for the dried trade, almost entirely for medicine (Vincent *et al*., 2011b). Research by Project Seahorse scientists estimated that millions of seahorses, predominantly dried, are likely traded each year (McPherson & Vincent 2004; Baum & Vincent 2005; Martin-Smith & Vincent, 2006; Giles *et al*., 2006; Perry *et al*., 2010; Vincent *et al.*, 2011a). The vast majority of seahorses in the dried trade appear to be taken from bycatch in trawl fisheries (McPherson & Vincent 2004; Baum & Vincent 2005; Giles *et al*., 2006; Perry *et al.*, 2010; Vincent *et al*., 2011a), but are obtained as bycatch in many other gears as well (Vincent *et al*., 2011a).

Concerns about the scale of the international trade in seahorses led to seahorses being listed in 2002 on Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). CITES is a multilateral environmental agreement currently including 180 member countries (representing 90% of the world's countries). CITES consists of three Appendices to the Convention; Appendix I essentially prohibits trade while Appendix III deals with taxa of national concern. Most species are listed under CITES Appendix II, which seeks to ensure that international trade of wild animals is sustainable. Appendix II is unusual in having legally binding mechanisms to discipline countries that do not comply with listings. The Appendix II listing for seahorses was implemented in 2004 and meant that all 180 Parties were required to monitor and regulate international seahorse trade through export and re-export permits, and prove that existing trade does not damage wild populations (CITES, 2004). Non-target fisheries that obtain seahorses present CITES Parties with a significant problem. Regulating exports may have little or no effect on seahorse capture rates when they are primarily obtained incidentally in bottom trawls (McPherson & Vincent, 2004; Baum & Vincent, 2005; Giles *et al*., 2006; Perry *et al*., 2010).

The life histories of seahorses may make them susceptible to population decline due to bycatch (Foster & Vincent, 2004). Some life history parameters of seahorses, including rapid growth, early sexual maturity, high natural mortality, short generation time, short life span, and year-round reproduction (Curtis & Vincent, 2006) are usually associated with opportunistic species, which are resilient to exploitation. Other life history traits, however, including intensive and specialized parental care, complex behavioural interactions, and monogamous pair bonds (common in many seahorse species) may make seahorses more intrinsically vulnerable to fishing impacts (Foster & Vincent, 2004). Further, fishing can generate depensatory effects in seahorse populations. Such effects take the form of, for example, (i) skewing sex ratios so that it becomes harder for seahorses in monogamous species to find a mate or (ii) removing larger individuals such that reproductive output declines (Vincent *et al*., 2011b).

The ecology of seahorses in terms or habitat use and population distribution can contribute to population decline. Their benthic habitat associations means that they are often retained in shrimp trawls; these vessels generally operate in shallow water habitats that are shared by both seahorses and target shrimp (Vincent *et al*., 2011b). Underwater surveys have reported that seahorse populations are naturally patchily distributed and rare (Foster & Vincent, 2004), which correlates to their patchiness and rarity in trawl catches (Giles *et al*., 2006; Perry *et al*., 2010). High vulnerability to habitat destruction by humans, population patchiness and rarity are all characteristics that increase vulnerability in marine species (Roberts & Hawkins, 1999; Reynolds *et al.*, 2005).

Bycatch of seahorses has only been investigated in isolated countries and regions (ie. Kenya and Tanzania: McPherson & Vincent, 2004; Latin America: Baum & Vincent, 2005; Vietnam: Giles *et al.*, 2006; Thailand and Malaysia: Perry *et al*., 2010) but has not been assessed on a global scale. Information on trawl fisheries gathered in these isolated countries and regions suggests that trawl gears obtain few seahorses per haul, but that these rare catches have the potential to scale up to many millions of seahorses being caught each year. Interviews with fishers that obtain seahorses in bycatch conducted in these countries and regions also suggest that catches of seahorses are declining. Such trends raise significant conservation concern for seahorses in bycatch, calling for more research into how the interaction between life history and bycatch impacts seahorse vulnerability.

1.3 Research objectives and thesis outline

My research was directed at answering four questions:

1. What are the life history parameters of three species of seahorses obtained as bycatch in peninsular Malaysia? (Chapter 2)
2. What do the life history parameters tell us about potential for overfishing for these three species of seahorses in peninsular Malaysia? (Chapter 2)
3. What gears obtain seahorses in bycatch and what are the parameters of those catches (Chapter 3)?
4. What can reported information on bycaught seahorse catch rates, catch volumes, and changes in catch tell us about the global fishing pressures on seahorses? (Chapter 3)

Chapter 2 uses life history-based approaches to assess fishing impacts on three bycaught species of seahorse in peninsular Malaysia. In peninsular Malaysia, significant volumes of seahorses in shrimp trawl bycatch had been noted during surveys conducted between 1998-1999 (Perry *et al*., 2010). No previous studies have used life history parameters to look for indicators of overfishing in seahorse catches.

Chapter 3 presents the first global review of seahorses in bycatch. Bycatch of seahorses in shrimp trawls throughout the developing world has been proposed as a major threat to seahorses (Vincent *et al*., 2011b), but there have been no global reviews of the issue to confirm this inference, or to identify other potential gear types of concern.

I end with a general discussion on the findings presented in this thesis and how they contributed to answering my research questions (Chapter 4).

1.4 Context and collaborations

The second chapter of this thesis focused on bycaught seahorses landed in peninsular Malaysia because I knew seahorses could be found in bycatch and because I developed vital working relationships with local colleagues. Previous research on seahorse landings (Choo & Liew, 2003) distribution (Lim *et al*., 2011) and trade (Choo & Liew, 2005; Perry *et al.*, 2010) in peninsular Malaysia provided important guidance for selecting landing sites to use in this field study. Other essential guidance on field research, logistics and culture came from partnerships that Project Seahorse developed with Dr. Chong Ving Ching and Adam Lim Chee Ooi at the University of Malaya. The memorandum of understanding between Project Seahorse/The University of British Columbia and the University of Malaya allowed me to conduct a four-month field survey under these colleagues' research permits in peninsular Malaysia (May – August 2013). This field survey was critical to the success of my second chapter.

The third chapter in this thesis relied heavily on a large collection of published and unpublished trade and fisheries surveys led by Project Seahorse researchers. Some of these data have been published (Baum *et al*., 2003; McPherson & Vincent, 2004; Baum & Vincent, 2005; Martin-Smith & Vincent, 2006; Giles *et al*., 2006; Meeuwig *et al*., 2006; Perry *et al*., 2010; Vincent *et al*., 2011a), and are widely available. Other data were unpublished and my ability and permission to access these unpublished reports played an important role in the data collection for my second chapter. Published and unpublished reports by several other researchers not directly associated with Project Seahorse helped to supplement my second chapter (Salin *et al*., 2005; Choo & Liew, 2005; Murugan *et al*., 2008; Alfaro-Shigueto & Mangel, 2011; Choi *et al*., 2012; Laksanawimol *et al*., 2013), but Project Seahorse researchers conducted the core research that was used to generate my global assessment of seahorses in bycatch.

Chapter 2: Novel life history data for threatened seahorses provides insight into fishing impacts

2.1 Introduction

The incidental capture of marine organisms has become a critical point of concern in fisheries management, as basic biological data for incidentally captured animals is often absent or sparse. The majority of studies on incidental capture have been focused on large or charismatic animals (Hall, 1997), yet the vast majority of animals that are incidentally caught are small fishes, maturing at lengths of less than 20 cm and weighing less than 100g (Alverson, 1994; Kelleher, 2005). These smaller fish species may be discarded or retained. Even those that are retained are seldom monitored, making species-specific assessment of fishing mortality and abundance challenging (Hall, 1997; Foster & Vincent, 2010b).

In response to a growing need to evaluate data-poor fisheries, life history-based approaches have been developed to discern the impacts of exploitation on marine fishes. Length measurements such as maximum body size (Dulvy & Reynolds, 2002) and size at maturity (Hutchings & Reynolds, 2004) have proven to predict intrinsic vulnerability to extinction in marine fishes effectively, with larger and later maturing species having a greater intrinsic vulnerability than those that are smaller and mature earlier (Winemiller, 2005). Froese (2004) used length at maturity to develop a data-poor method to assess overfishing, by applying three simple indicators to catch data. Catch data can also provide insight into population-level effects of fishing. For example, sex bias in commercial catches predicts a higher rate of population decline and a greater time needed for recovery following collapse (Rowe & Hutchings, 2003). Although most data-poor assessment techniques are aimed at using sparse information to make an assessment of a fishery, they still often require information such as a time series of catches; or estimates of fishing mortality, biomass or age at maturity (Carruthers *et al.*, 2014), which may not be confidently known for some species.

Seahorses (*Hippocampus* spp.) provide examples of incidentally captured small fish where basic life history is available for very few of the species that are being affected by fisheries. Currently 75% of the *c*. 48 species of seahorse are listed as ‘Data Deficient’ on the International Union for Conservation of Nature (IUCN) Red List, meaning that limited data has prevented a reliable assessment of extinction risk for these species (IUCN, 2013). Those studies that have investigated seahorse life history characterized them as having early sexual maturity, high natural mortality, a short life span and rapid growth relative to other teleosts (Curtis & Vincent, 2006; Harasti *et al*., 2012). Yet other traits like complex social interactions, including parental care and monogamous mating patterns, may make seahorses more likely to be overfished (Sadovy, 2001; Foster & Vincent, 2004). Discovering and reconciling the mixture of seahorse life history traits is critical to understanding the impacts of their incidental capture.

It is estimated that the international trade in seahorses, fueled by demand for traditional Chinese medicine, the aquarium trade and curios, obtains 95% of its seahorses from incidental capture (Vincent *et al.*, 2011b). The 2002 addition of seahorses to Appendix II of the Convention for International Trade in Endangered Species (CITES) has generated formal data on legal and reported international trade (Vincent *et al.*, 2013). Yet little is known about the role of incidental capture in such international trade or how it is affecting seahorse populations. It is known, however, that the five data-poor species of seahorse that account for 91% the reported international trade are frequently obtained through incidental capture (Vincent *et al*., 2011b; UNEP-WCMC, 2012; Foster *et al*., in revision).

This study focuses on Malaysia, a country that historically exported four of the five seahorse species that dominate international trade, and was the fifth largest global seahorse-exporting nation between 2004 and 2011 as reported to CITES (UNEP-WCMC, 2012; Foster *et al*., in revision). A trade survey carried out in 1998/99 estimated that the Malaysian trawl fishery alone obtained one million seahorses (approximately 2900 kg) annually through incidental capture (Perry *et al*., 2010). Previous studies on seahorses in peninsular Malaysia have focused on reporting species distribution (Choo & Liew, 2004; Lim *et al*., 2011) and examining trade routes (Choo & Liew, 2005; Perry *et al*., 2010), but none have published life history information on the three focal species in this study.

This study had two goals – the first was to generate critical information on seahorse life history that could aid in employing assessment models for data-poor fisheries. The second was to apply this novel information to seahorse incidental capture in peninsular Malaysia, with the aim of identifying vulnerable seahorse populations. Results from this study will be applicable to surrounding Southeast Asian nations, whose non-selective fisheries catch the same seahorse species.

2.2 Materials and methods

2.2.1 Study species

This study focuses on three seahorse species in peninsular Malaysia: Kellogg's seahorse *Hippocampus kelloggi* (Jordan & Snyder 1901), the hedgehog seahorse *Hippocampus spinosissimus* (Weber 1913) and the three-spot seahorse *Hippocampus trimaculatus* (Leach 1814), that were identified according to Lourie *et al.* (2004). All three species are distributed throughout the Indian and western Pacific Oceans and are associated with soft bottom and coral habitats (Foster & Vincent, 2004). Knowledge of the life history parameters for these three species were either completely unknown (*H. kelloggi*) or only partially described (*H. spinosissimus*: Meeuwig *et al*., 2006; *H. trimaculatus*: Meeuwig *et. al*., 2006; Murugan *et. al*., 2009; Murugan *et al.*, 2011). Together, the three species made up 75% of trade volumes that were reported to CITES from 2004-2011 (annual range by number of individuals): *H. trimaculatus* (1.1-2.5 million); *H. spinosissimus* (1.0-2.5 million); and *H. kelloggi* (0.75-1.4 million; UNEP-WCMC, 2012; Foster *et al*., in revision).

2.2.2 Fisheries-dependent surveys

Catches were haphazardly surveyed for seahorses at 31 landing sites found along the west coast of peninsular Malaysia (from 06°06'N, 100°17'E to 01°19N, 103°26E) and at two landing sites on the southeast coast of peninsular Malaysia (from 02°26'N, 103°49'E to 02°39'N, 103°36'E). The northeast coast of peninsular Malaysia was not visited during this study due to logistical constraints. Sites were visited from May to August 2013 as shown in Figure 2.1a. The landing sites surveyed in this study ranged from small fishing villages to large, government-operated ports.

During the first month of surveys in May 2013, 31 landing sites (all located on the west coast of peninsular Malaysia) were visited, with seahorses encountered in 22 of these (Figure 2.1a). Fishers often had stockpiles of dried seahorses, and all seahorses that were encountered were measured. Although these stockpiles were treated as random samples of the fished population, fishers may be more likely to retain large seahorses that are higher in value (Perry *et al*., 2010). During this initial survey period, 361 stockpiled seahorses from 26 fishers at 22 landing sites were measured. Fishers reported that these seahorses had been collected within the previous two-year period (median = nine months, n = 26 fishers). Fishing gears used by these fishers included bottom trawl nets, bag nets, tidal shrimp nets (gombang 'Y-nets'), drift nets and purse seines.

Fishers were interviewed to identify landing sites where seahorses were frequently caught. Those who claimed to catch seahorses were asked if they would be willing to collect seahorses as part of this research project. In total, ten fishers agreed to assist with this project: five fishers (from four landing sites) collected dried seahorses, and five fishers (from five landing sites) agreed to collect freshly dead seahorses in 95% ethanol. Participating fishers were visited at one-month intervals from June to August 2013 to measure seahorse collections (Figure 2.1a). These monthly visits to participating fishers resulted in a total of 486 seahorses (112 in ethanol and 374 dry).

Seahorse collectors, who purchased seahorses from fishers, were interviewed at three landing sites (Figure 2.1a). Four seahorse collectors allowed us to measure 307 dried seahorses in total. Three of these collectors were located at two landing sites on the east coast of peninsular Malaysia, a region that was only visited in July 2013 and was not visited monthly. The fourth collector was located on the west coast, and was visited monthly from June to August.

Data collection involved recording the height, mass, sex, reproductive state and maturity of each individual seahorse. Seahorse 'height' was measured as the distance from the tip of the coronet to the tip of the outstretched tail (Lourie, 2003). Seahorse mass was taken using a Chestnut Tools pocket scale that was accurate to the nearest 0.1 g. Male seahorses were visually distinguished from females by the presence of a brood pouch or, for juvenile males, the presence of a darkened oval zone where a brood pouch was developing (Boisseau, 1967).

For mature males, reproductive status was assigned based on categories developed by Perante and colleagues (2002) as stage 0 (pouch empty and taut), stage 1 (pouch slightly distended or newly empty) or stage 2 (heavily pregnant, where the pouch was very distended). Animals that were physically mature (had a brood pouch), however, were not necessarily reproductively active (engaging in reproduction), as noted in an *in situ* study by Harasti *et al.* (2012). To examine this difference, males of reproductive stages 0, 1 and 2 were considered to be physically mature, but only those of reproductive stages 1 and 2 were considered to be reproductively active, in accordance with Morgan & Vincent (2013). As female maturity state can only be determined by dissecting ovaries in freshly dead or preserved specimens (Foster & Vincent, 2004), and sampling obtained just six females in ethanol, females were necessarily assumed to mature at the same size as males.

Given the temporal constraints of this study period, it was assumed that all three species displayed a year-round breeding season, in accordance with the majority of tropical seahorses as described in Foster and Vincent (2004). Indeed studies on *H. spinosissimus* and *H. trimaculatus* in Vietnam also confirmed year round breeding seasons but with peaks in reproductive activity at certain times of the year (Meeuwig *et al*., 2006).

2.2.3 Data analyses

Seahorse samples that were acquired were either dried or preserved in 95% ethanol. In order to account for shrinkage that occurs during the drying process, and when stored in ethanol, the heights of preserved samples were mathematically converted to dried heights using equations provided in Nadeau *et al*. (2009). Mean heights were not significantly different when ethanol preserved sample heights were mathematically converted to dry-equivalent sample heights for *H. spinosissimus* (Mann-Whitney *U*-test, W = 250345, $P = 0.89$) or for *H. trimaculatus* (Mann-Whitney *U*-test, W = 21173, $P = 0.37$). Therefore both dried and preserved samples were pooled in subsequent analysis. All samples of *H. kelloggi* were dried, so conversion was not necessary.

The height at physical maturity (Ht_m) was determined by fitting a logistic regression to the proportion of mature males (reproductive stages 0, 1 and 2) out of all observed males in a given 10 mm height class to calculate the 50% transition point (King, 2007). The logistic regression was fitted using a non-linear least squares search function in the R statistical platform (R Development Core Team, 2013). Similarly, height at reproductive activity (Ht_r) was determined by fitting a logistic regression to the proportion of reproductively active males (reproductive stages 1 and 2) in a given 10 mm size class against all males in that size class (in accordance with Morgan & Vincent, 2013).

Minimum, maximum and mean heights were determined for each seahorse species examined in this study. The relationship between mass and height was also determined for each species by fitting a log-linear regression to these data. Sexes were pooled for each species because the height-mass relationship was found not to differ between the sexes (*H. kelloggi* ANCOVA, F=1087, d.f. = 95, $P = 0.95$; *H. trimaculatus* ANCOVA, F= 50.46, d.f. = 183, $P = 0.23$; *H. spinosissimus* ANCOVA, F=192.1, d.f. = 116, $P = 0.29$). To test for evidence of sexual dimorphism, mean heights were compared between sexes using the Mann-Whitney *U*-test, as samples were not normally distributed (*H. kelloggi* Shapiro-Wilk Normality Test, W = 0.9489, $P < 0.001$; *H. spinosissimus* Shapiro-Wilk Normality Test, W = 0.994, $0.05 > P > 0.01$; *H. trimaculatus* Shapiro-Wilk Normality Test, W = 0.9423, $P < 0.001$). Height-mass relationships were examined using analysis of co-variance with sex or region as the covariate, as log-transformed sample distributions were approximately normal. Comparisons between the sexes were carried out for mature animals only. Where maturity was ambiguous, a category of maturity was assigned retroactively based on estimated height at physical maturity for males **(**Ht_m) for each species (see Maturity below). Sex ratio was calculated as the proportion of mature individuals that were males. A chi-square test was used to identify if the ratio was significant different from unity. Data analyses were carried out using the R statistical platform (R Development Core Team, 2013).

Height at first capture (Ht_c), the point at which a given seahorse had a 50% chance of being retained when it encountered a fishing net (Sparre *et al*., 1989), was calculated by fitting a logistic regression to the proportion of individuals caught in a given 10 mm height class compared to the size class where the most individuals were retained. These analyses were completed using a non-linear least squares search function in the R statistical platform (R Development Core Team, 2013).

Spatial patterns in mean height were examined among the regions identified in Figures 2.1a and 2.1b where data was available, and temporal patterns were examined using only those samples for which the month of capture was known (June, July or August). As sample sizes allowed, spatial and temporal differences were analyzed with respect to mean height, height-mass relationship, sex ratio, height at first capture (Ht_c), and height at maturity (Ht_m). Spatial patterns could only be examined for *H. spinosissimus* and *H. trimaculatus*, as all *H. kelloggi* samples were obtained from a single location. Spatial and temporal analysis was completed using Kruskal-Wallis tests to compare mean heights, height-mass relationships, height at first capture (Ht_c), and height at maturity (Ht_m). Chi-square tests were used to compare sex ratios. All data analyses were carried out using the R statistical platform (R Development Core Team, 2013). Tests were considered significant when P-values were <0.05.

2.3 Results

2.3.1 Distribution of seahorses in catches

Seahorses were encountered at 24 of 33 landing sites surveyed in this study (Figure 2.1a), with varying distribution by species (Figure 2.1b). *Hippocampus trimaculatus* was the most commonly sampled species of seahorse, encountered at all 23 landing sites with seahorses (Figure 2.1b). The second most commonly encountered species was *H. spinosissimus*, which was sampled from four landing sites: two on the southeast coast and two on the southwest coast (Figure 2.1b). *Hippocampus kelloggi* was found in just one landing site on the west coast of the peninsula.

2.3.2 Maturity

Estimates for height at 50% physical maturity (Ht_m) were obtained for *H. spinosissimus* and *H. trimaculatus* (Table 2.1, Figure 2.2b and c respectively), but not for *H. kelloggi* as only one of the sampled males was immature (Table 2.1, Figure 2.2a). Height at 50% reproductive activity (Ht_r) was modeled for all three species (Table 2.1, Figure 2.2a-c). Samples were necessarily pooled across the peninsula by species to allow for a robust estimate of height at 50% maturity.

2.3.3 Gear retention

Height at 50% capture (Ht_c) was largest for *H. kelloggi* and smallest for *H. trimaculatus* (Table 2.1, Figure 2.3a and c respectively). For comparison between gear retention and maturity, histograms showing the range of all sampled heights with Ht_r, Ht_m, Ht_c indicated for each species are presented in Figure 2.4.

2.3.4 Mean height

Mean height varied among species, and between sexes for two of the three species sampled in this study. *Hippocampus kelloggi* had the largest mean height (169.3mm ± 34.4) and *H. trimaculatus* had the smallest (96.8 mm ± 15.8; Table 2.2). Males were larger than females for *H. spinosissimus* and for *H. trimaculatus*, but no significant difference in sampled mean height was found between sexes for *H. kelloggi* (Table 2.2).

Spatial and temporal differences in mean height were also found. Mean height of the samples obtained in this study differed across regions for *H. trimaculatus*, but not for *H. spinosissimus*. Mean height of *H. trimaculatus* samples were smallest in the southwest region when compared to the northwest and southeast regions (Table 2.3). *Hippocampus kelloggi* and *H. spinosissimus* both showed significant differences in mean height across the three sampling months. For *H. kelloggi* the mean height of samples collected in June was significantly smaller than the mean heights observed in July and August, and for *H. spinosissimus* the mean height of samples collected in August were significantly smaller than those collected in June and July. For *H. trimaculatus* differences could only be compared between June and July because of sample size limitations, and no significant difference was found (Table 2.3).

2.3.5 Sex ratio

The ratios of sampled mature males to females for *H. kelloggi*, *H. spinosissimus* and *H. trimaculatus* were not significantly different from a 1:1 ratio (Table 2.2). Only *H. trimaculatus* showed a spatial pattern in sex ratio, with more mature females sampled than mature males in the southwest region of peninsular Malaysia (6% males, d.f. = 1, $\chi^2 = 24.5$, $P < 0.001$).

2.3.6 Height-mass relationships

The log-linear relationship between height and dry mass was significant for all three species (Table 2.4). Although significant, height only predicted 59% of the variation in mass for *H. trimaculatus* when all samples were pooled. This was due to the poor condition of some of the dried samples, and was improved when analysis was carried out using preserved animals only. For *H. spinosissimus* only, seahorses sampled from the northwest coast were significantly heavier than those from the southeast coast (northwest coast slope and 95% C.I. = 3.2 (3.0 – 3.5), $r^2 = 0.92$, $P < 0.001$; versus southeast coast slope and 95% C.I. = 2.8 (2.6 – 3.1), $r^2 = 0.87$, $P < 0.001$; ANCOVA, F = 4.565, d.f. = 181, $0.05 > P > 0.01$).

2.4 Discussion

This study generates critical *in situ* life history data for three heavily traded species of seahorse *H. kelloggi*, *H. spinosissimus* and *H. trimaculatus*, and applies this data to rapidly assess the potential for overfishing of these three species in peninsular Malaysia. When samples were from peninsular Malaysia were pooled, capture occurred at approximately the same height as physical maturity and sex ratios were equal for all species, which might have suggested little reason for concern. However, because height at first capture was less than height at actual reproductive activity for all species, many individuals appeared to become retained in gear prior to having the opportunity to reproduce. Additionally, regional analyses raised concern for *H. trimaculatus*, the only species sampled from the southwest of the peninsula, as these samples had traits that are often associated with overfished populations, including a highly skewed sampled sex ratio and a smaller mean body size.

Comparing size at maturity to size at which 50% of specimens are retained in fishing gear is a common data-poor assessment method for teleosts (e.g. Rueda & Defeo, 2003; Foster &

Vincent, 2005; Kuparinen *et al*., 2009; Foster & Vincent, 2010b). For seahorses, however, physical maturity is not always indicative of behavioural or physiological maturity (Cai *et al*., 1984; Harasti *et al.*, 2012; Morgan & Vincent, 2013). Two estimates of height at maturity – that of physical maturity and that of reproductive activity – provided strikingly different insight into the sustainability of the catches. Height at first capture was either similar to (*H. spinosissimus* and *H. trimaculatus*) or greater than (*H. kelloggi*) height at physical maturity for all species. This could suggest that, on average, these species are able to reproduce before becoming vulnerable to capture. However, when height at first capture was compared to height at reproductive activity, all three species were captured well before (~20-40 mm) they actually started reproducing and therefore were unable to contribute to the next generation. In another species of seahorse, 20 mm of growth corresponded to 1.5 months, a time period that is likely critical for pair bonding (Morgan & Vincent, 2013). Therefore, the common practice of using external indicators of maturity for seahorses (*H. kelloggi*: Lourie *et al*., 1999; *H. trimaculatus*: Murugan *et al.*, 2009; *H. spinosissimus*: Nguyen & Do, 1996) would potentially underestimate fishing impact. Perry *et al*., (2010) noted that larger seahorses in peninsular Malaysia fetched higher prices; therefore it seems possible to assume that fishers may discard smaller, potentially less valuable individuals. If this were the case, fishing impact would be underestimated, as it would lead to an overestimation of height at 50% capture.

This distinction between physical maturity and reproductive activity is important as size at maturity plays a critical role in many data-poor assessment techniques. For example, Froese (2004) created three indicators that rely solely on an estimate of length at maturity to assess a catch for the percent that is: (a) over length at maturity, (b) within an optimal length-weight range and (c) considered to be mega-spawners. Using physical maturity to assess the first three of these indicators (a) for *H. trimaculatus* catch in peninsular Malaysia would indicate that 61% of the catch was mature, but only 5% would be considered mature when using height at reproductive activity instead. This is concerning, as the majority of the catch may be removed during an important pair bonding period (Morgan & Vincent, 2013). In contrast, for *H. spinosissimus,* 85% of the catch was above height at physical maturity, and 59% above height at reproductive activity, which is far less concerning. These examples show that data-poor assessment techniques based strictly on height at maturity may provide very different views on the health of the catches, depending on how maturity is defined.

The distribution of species in samples varied spatially, and were consistent with results of another fisheries-dependent survey conducted in 2001 in peninsular Malaysia (Choo & Liew, 2003), yet the reasons for the observed distribution are not fully understood. *Hippocampus spinosissimus* was not encountered in the southwest even though *H. trimaculatus* was, and these two species are thought to share habitat and depth preferences (Choo & Liew, 2003; 2004). The reason for why *H. spinosissimus* was absent from the southwest is unclear. The spatial differences in sample distribution observed here cannot be equated with true species distribution due to differences in fishing effort and behaviour across the peninsula (Hiew *et al.*, 2013). However, the distribution of most seahorse species is so poorly understood (Foster & Vincent, 2004) that information obtained through fisheries-dependent surveys can still provide valuable insight for seahorses, even when the reasons for observed distribution patterns are not fully understood.

The peninsular-wide catches of *H. trimaculatus* raised the greatest concern among the three species when assessed using height at reproductive activity. In particular, samples in the southwest had a strong female-biased sex ratio in catches, and a smaller mean height compared to the northwest and northeast regions of the peninsula. A female-biased sex ratio has been found in other fished populations of seahorses (ie. *H. erectus*: Baum *et al*., 2003), and in seahorses sampled from areas with poor environmental conditions (ie. *H. erectus* in Chesapeake Bay: Teixeira & Musick, 2001). Previous studies on the incidental capture of *H. trimaculatus* have reported both equal sex ratios (Vietnam: Meeuwig *et al.*, 2006), and female-biased sex ratios (India: Murugan *et al.*, 2011). Skewed ratios could also be explained by a violation of the assumption that females mature at the same size as males, but then one would expect to see skewed ratios for all regions, and not just in one region as was the case for *H. trimaculatus* in this study. Skewed sex ratios may also be associated with spatial segregation between males and females in the population – thus affecting catchability – or it may be indicative of disruption to the breeding population through fishing impacts (Baum *et al*., 2003). Heavy fishing pressure has been shown to reduce mean size in other fish species by selecting for individuals that reach maturity earlier (Hutchings & Baum, 2005), and such fishing pressure may be responsible for the smaller mean height of *H. trimaculatus* observed in the southwest region. These results suggest potential for disruption to the breeding population of *H. trimaculatus* in southwest peninsular Malaysia, as a highly skewed sex ratio reduces the amount of potential mates (Rowe & Hutchings, 2003), and brood size has been shown to increase with male body size in other species of seahorse (Teixeira & Musick, 2001).

A lack of time-series data from long-term monitoring makes it impossible to conclusively determine if seahorses in the southwest are overfished, yet the results from this study can provide valuable information for future assessment. The southwest region of Malaysia has been undergoing rapid development, and damage to coastal ecosystems due to development is of top concern to the local communities (Lim *et al*., 2011). These results provide direction to seahorse research efforts in the region. In order to develop robust assessments of these populations future efforts should focus on estimating abundance and fishing mortality (Walters & Martell, 2004) and the spatial distribution of each species.

The three heavily traded seahorse species examined in this study demonstrate that, in the face of limited data, simple catch information can be used to estimate life history parameters and identify populations that may be overfished. While this study provides the first robust *in situ* estimates of life history parameters for *H. kelloggi*, and is among the first to do so for *H. spinosissimus* and *H. trimaculatus*, the methods described are applicable to many species of small, incidentally captured fishes. Small fishes are often considered to be less vulnerable to overfishing (Jennings *et al*., 1999), yet this study and others demonstrate that these fishes too can show signs of overfishing (ie. Foster & Vincent, 2010b). The results in this study also showed that overfishing assessments for these species can differ based on how length at maturity is defined, a parameter that is often the cornerstone of data-poor assessment methods. Ideally, length at maturity should always be defined as the point where reproductive activity begins. This is especially true for rare or endangered species where maturity is often determined externally (e.g. skates and rays: Estalles *et al*., 2011; fin whales: Aguilar & Lockyer, 1987; leatherback turtles Stewart *et al*., 2007).

Table 2.1 Values for height at 50% capture (Ht_c), height at 50% physical maturity (Ht_m), and height at 50% reproductive activity (Ht_r) and the associated 95% confidence intervals, reported in millimetres, for sampled male *Hippocampus kelloggi* (n= 48), *H. spinosissimus* (n= 77) and *H. trimaculatus* (n= 197). These data were obtained from May to August 2013 across peninsular Malaysia. Also included are the γ values associated with the logistic curve, reported with 95% confidence intervals.

	H. kelloggi		*H. spinosissimus*		*H. trimaculatus*	
	Mean	95% C.I.	Mean	95% C.I.	Mean	95% C.I.
Ht_c	132.5^1	127.3 – 137.7	106.0^1	101.1 – 110.9	82.5^1	82.1 – 82.9
γ	0.14^3	0.04 – 0.25	0.10^3	0.05 – 0.15	0.29^1	0.26 – 0.32
Ht_m	N/A	N/A	99.6^1	94.2 – 105.0	90.5^1	90.2 – 90.8
γ	N/A	N/A	0.09^3	0.04 – 0.14	0.16^1	0.16 – 0.17
Ht_r	167.4^1	148.8 – 186.0	123.2^1	117.0 – 129.4	121.8^1	116.1 – 127.5
γ	0.04^3	0.01 – 0.06	0.08^2	0.04 – 0.11	0.06^2	0.04 – 0.08

P-values from logistic regressions: $^1P < 0.001$, $^2\ 0.01 > P > 0.001$, $^3\ 0.05 > P > 0.01$

Table 2.2 Mean height (mm), range of height (mm), standard deviation from the mean (mm) and sample size (number of individuals) for mature males (M), mature females (F), and all samples (including juveniles) for *Hippocampus kelloggi, H. spinosissimus* and *H. trimaculatus*. These data were obtained from May to August 2013 across peninsular Malaysia. P-values and test statistics from Mann-Whitney *U*-tests comparing mean heights of mature males to mature females are reported, as well as sampled sex ratios (males to females) using mature animals only. Reported p-values for sex ratios were obtained using a χ^2 test.

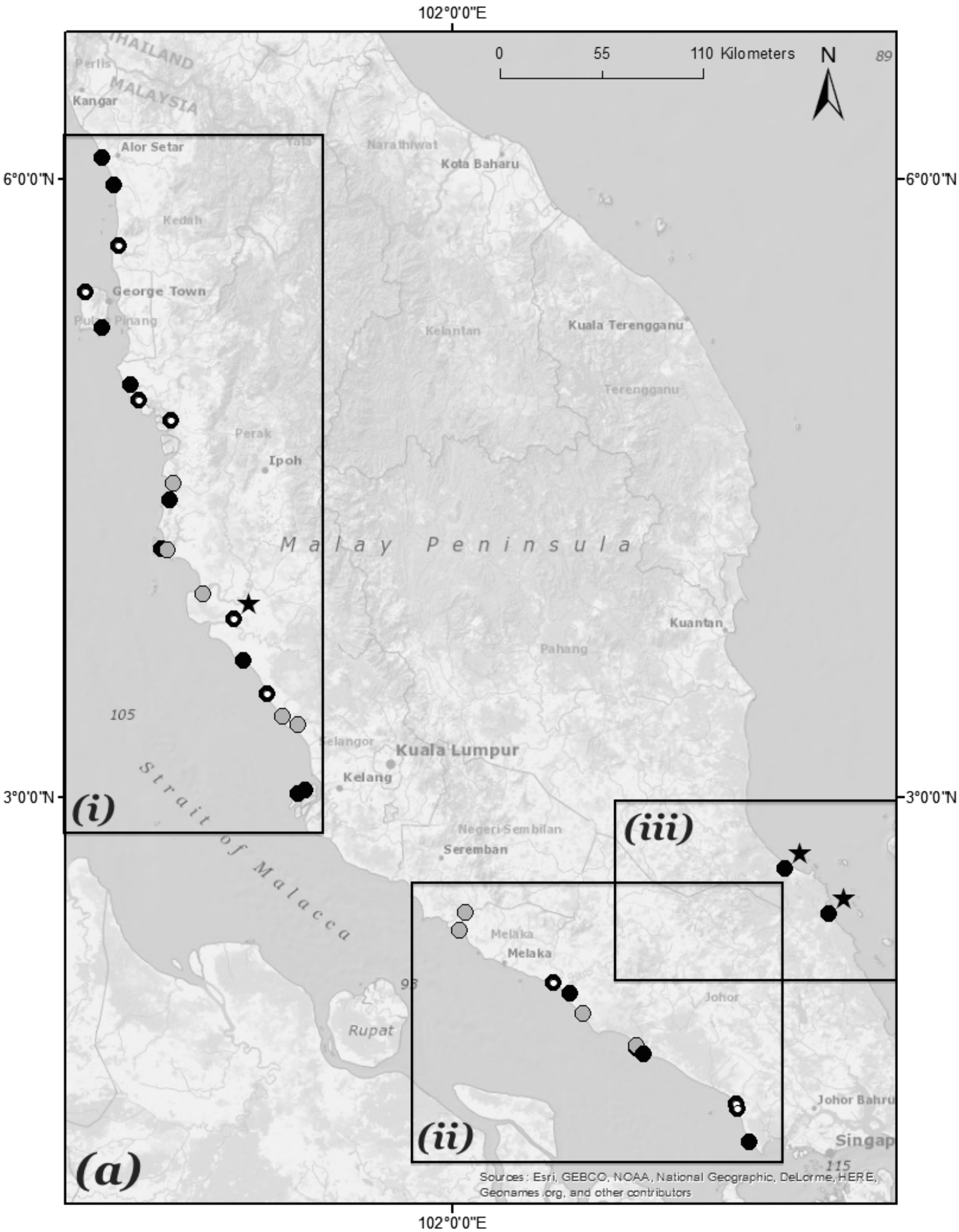
	Sex	Mean Height	SD	Min Height	Max Height	Sample size	Height P-value and test statistic	Sex Ratio and P-value
H. kelloggi	All	169.3	34.4	101	246	100		
	M	171.9	32.3	124	246	48	$P = 0.43$	0.48
	F	167.5	36.6	101	230	52	W = 1133	($P = 0.69$)
H. spinosissimus	All	126.4	20.9	70	175	205		
	M	135.4	16.5	101	175	77	$0.05 > P > 0.01$	0.44
	F	129.4	16.7	100	169	97	W = 2956	($P = 0.13$)
H. trimaculatus	All	96.8	15.8	48	175	688		
	M	107.0	12.8	91	175	197	$P < 0.001$	0.47
	F	103.8	13.7	91	164	224	W = 16921	($P = 0.22$)

Table 2.3 Temporal (June, July and August) and spatial (northwest, southwest and southeast regions) mean heights and associated 95% confidence intervals, and sample sizes (n) for *Hippocampus kelloggi*, *H. spinosissimus*, and *H. trimaculatus*. These data were obtained from May to August 2013 across peninsular Malaysia. Significant spatial and temporal differences in mean height are indicated by *. (P-values <0.001 from Kruskal-Wallis or Mann-Whitney *U* tests).

	H. kelloggi		*H. spinosissimus*		*H. trimaculatus*	
	Mean (n)	95% C.I.	Mean (n)	95% C.I.	Mean (n)	95% C.I.
			Temporal			
Jun	154.7* (59)	147.0-162.3	129.8 (80)	125.5-134.0	97.4 (180)	95.3-99.5
Jul	198.5 (22)	185.9-211.2	137.2 (32)	131.9-142.4	98.4 (140)	96.3-100.5
Aug	176.7 (17)	165.8-187.7	110.9* (24)	101.4-120.4	N/A	N/A
			Spatial			
NW	N/A	N/A	126.0 (161)	122.8-129.2	97.5* (513)	96.5-98.6
SW	N/A	N/A	N/A	N/A	86.2* (149)	84.5-88.1
SE	N/A	N/A	128.0 (44)	121.6-134.4	141.7* (26)	134.9-148.6

Table 2.4 Relationship between height (Ht, mm) and mass (M, nearest 0.1g) as fitted by $M=a*Ht^b$ (where a is the intercept and b is the slope of the line) for *Hippocampus kelloggi, H. spinosissimus* and *H. trimaculatus*. Samples were pooled from May to August 2013, and were obtained across peninsular Malaysia. Probability (P-values) indicates significance of r^2 values.

	Sample condition	d.f.	b	a	r^2	*P*
H. kelloggi	Dry	96	3.15	$6.56*10^{-7}$	0.958	< 0.001
H. spinosissimus	Dry	182	3.14	$7.80*10^{-7}$	0.867	< 0.001
H. trimaculatus	Dry	666	2.41	$1.35*10^{-5}$	0.590	< 0.001
H. trimaculatus	Ethanol	67	3.55	$1.39*10^{-7}$	0.842	< 0.001


102°0'0"E
0 55 110 Kilometers
N
THAILAND
MALAYSIA
Perlis
Kangar
Alor Setar
Narathiwat
Kota Baharu
6°0'0"N
Kedah
George Town
Pulau Pinang
Kelantan
Kuala Terengganu
Terengganu
Perak
Ipoh
Malay Peninsula
Kuantan
Pahang
105
Selangor
Kuala Lumpur
Kelang
3°0'0"N
(i)
Strait of Malacca
Negeri Sembilan
Seremban
(iii)
Melaka
Johor
Rupat
Johor Bahru
Singap
115
(ii)
(a)
Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, HERE, Geonames.org, and other contributors

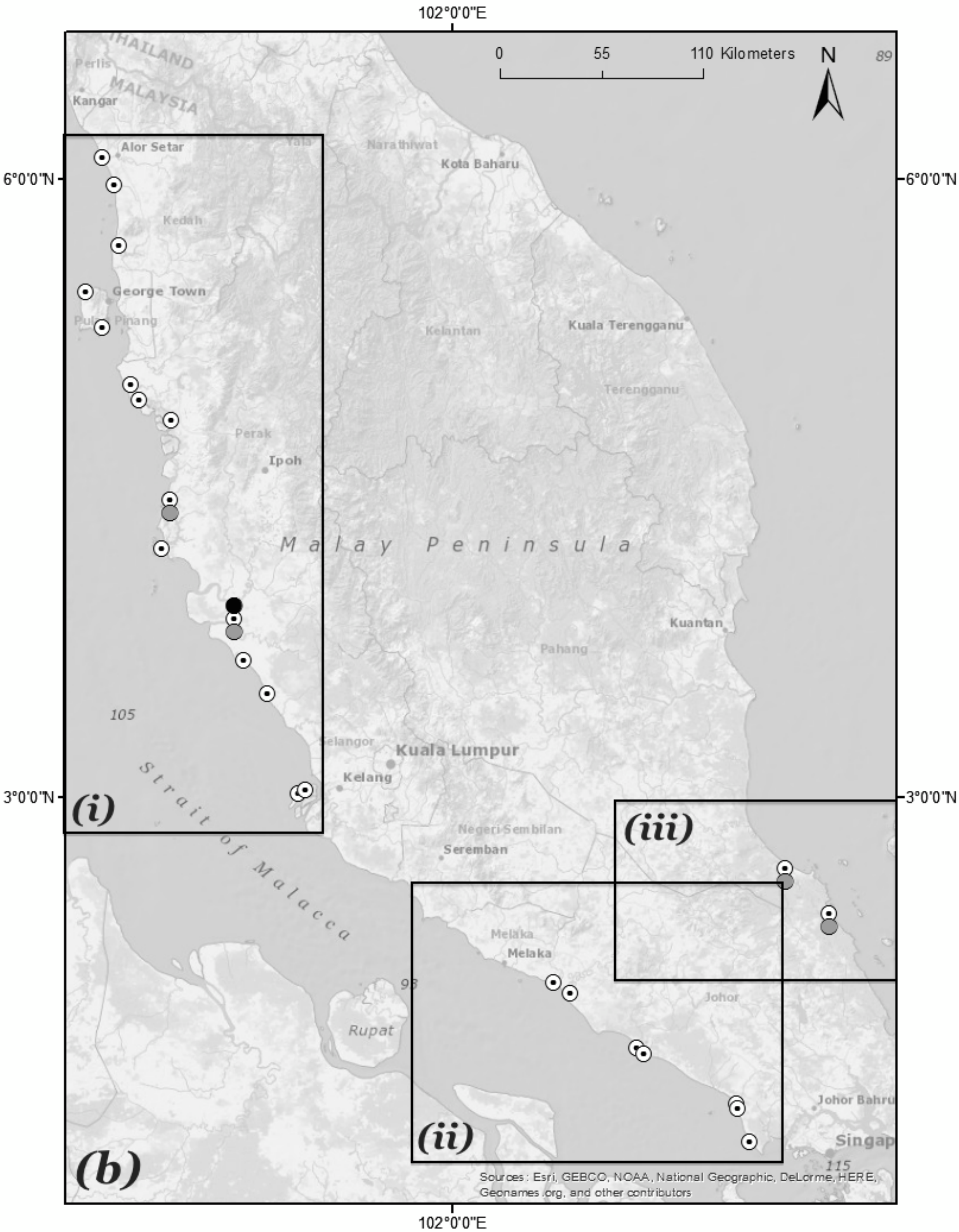

102°0'0"E
0 55 110 Kilometers
N
6°0'0"N
3°0'0"N
THAILAND
MALAYSIA
Perlis
Kangar
Alor Setar
Yala
Narathiwat
Kota Baharu
Kedah
George Town
Pulau Pinang
Kelantan
Kuala Terengganu
Terengganu
Perak
Ipoh
Malay Peninsula
Kuantan
Pahang
Selangor
Kuala Lumpur
Kelang
Negeri Sembilan
Seremban
Strait of Malacca
Melaka
Rupat
Johor
Johor Bahru
Singap
(i)
(ii)
(iii)
(b)
Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, HERE, Geonames.org, and other contributors

Figure 2.1 (a) All the landing sites visited throughout peninsular Malaysia, indicating the presence (●) or absence (○) of seahorse samples from fishers at these sites. Also indicated are the landing sites sampled monthly (◉) and sites where additional samples were obtained from seahorse collectors (★). (b) The distribution of *Hippocampus kelloggi* (●), *H. spinosissimus* (○) and *H. trimaculatus* (⊙) found in samples. Boxes indicate the Northwest (i), Southwest (ii) and Southeast (iii) regions as described throughout the analysis. Maps were generated using ESRI ArcGIS 9.3.

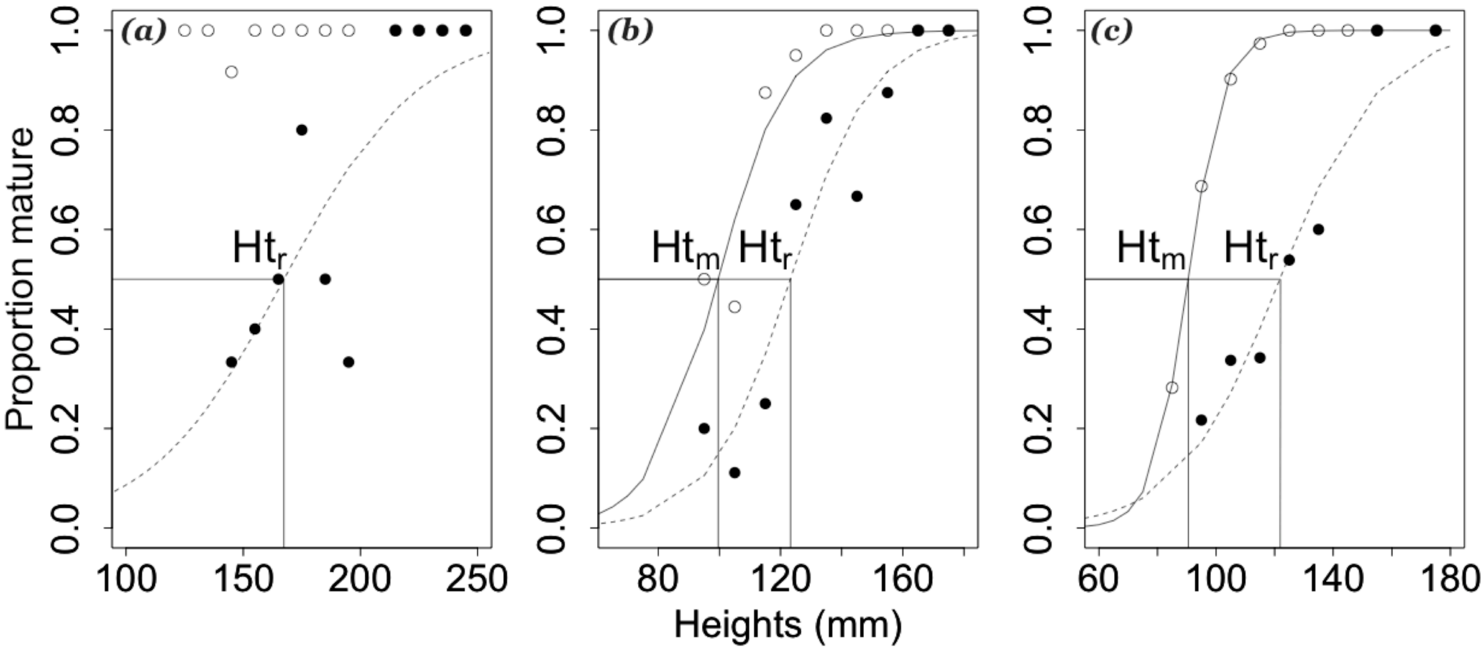


Figure 2.2 Logistic regression curves showing height at 50% male maturity (Ht_m, ●) and height at 50% male reproductive activity (Ht_r, ○) for (a) *Hippocampus kelloggi,* (b) *H. spinosissimus* and (c) *H. trimaculatus* obtained from May to August 2013 across peninsular Malaysia. For *H. kelloggi*, only Ht_r could be fit to a logistic regression as all observed individuals were mature.

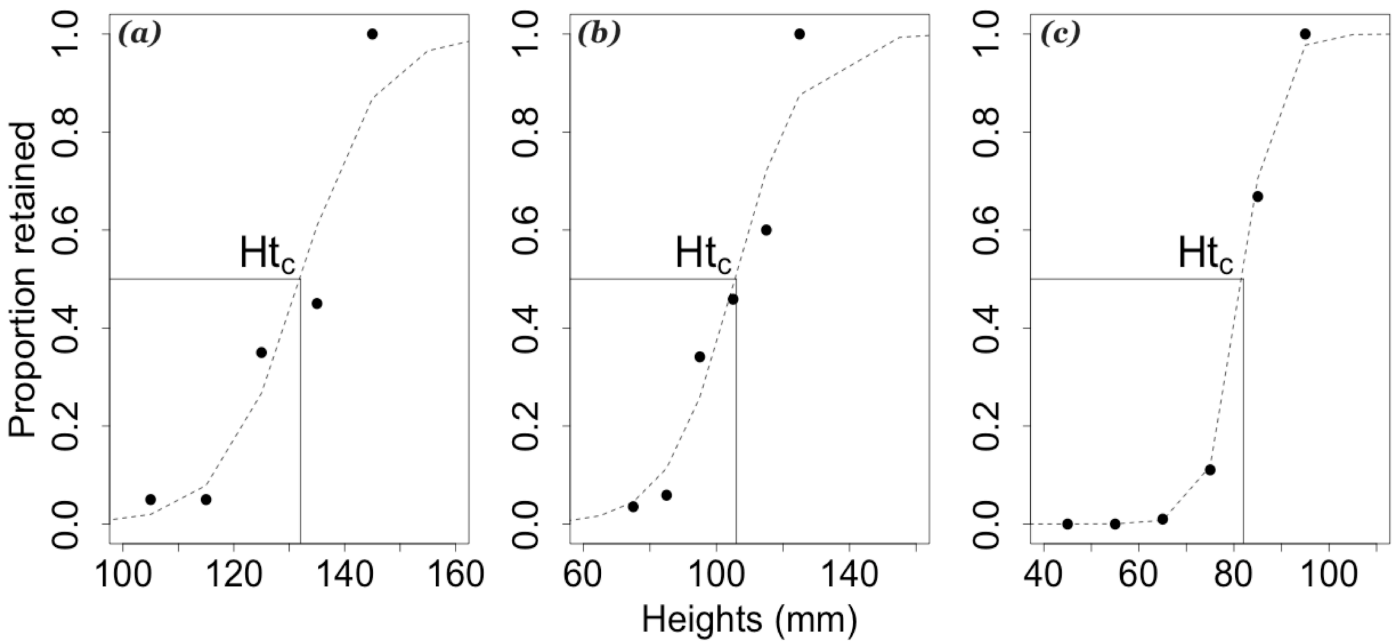


Figure 2.3 Logistic regression curves showing the height at first capture (Ht_c), the point at which a given seahorse has a 50% chance of being retained in a fishing net, for (a) *Hippocampus kelloggi,* (b) *H. spinosissimus* and (c) *H. trimaculatus* obtained from May to August 2013 across peninsular Malaysia.

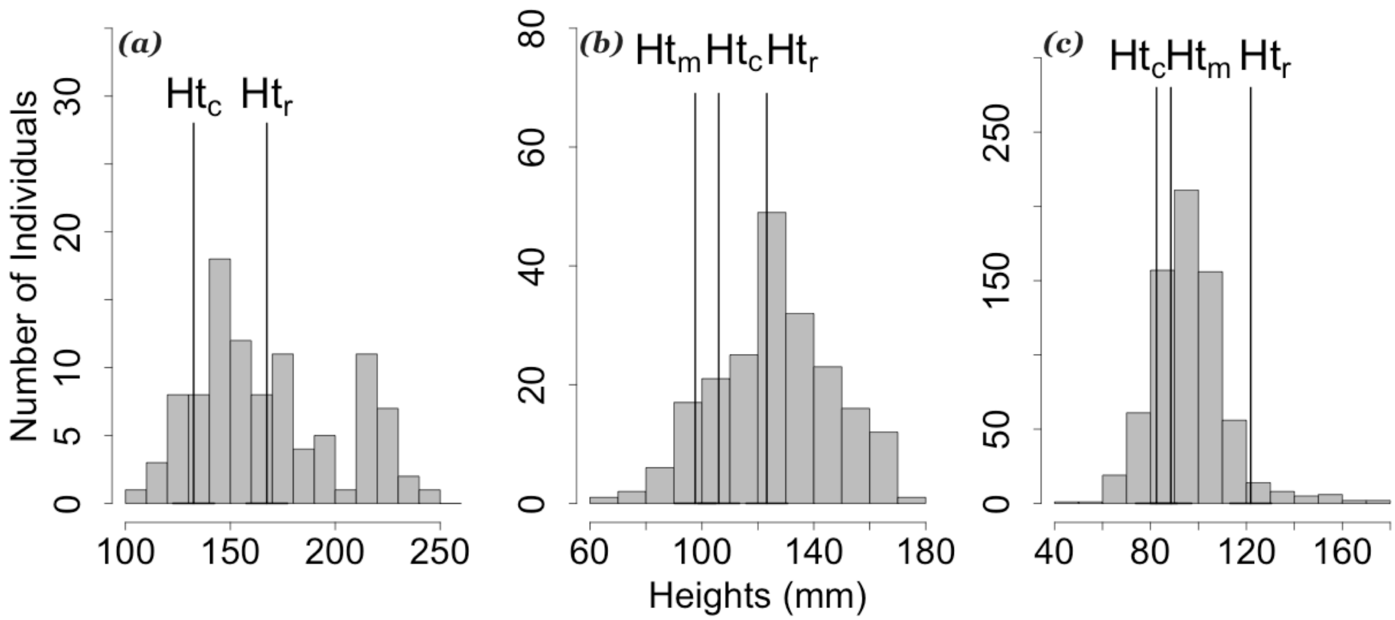


Figure 2.4 Total height frequency histograms for (a) *Hippocampus kelloggi*, (b) *H. spinosissimus* and (c) *H. trimaculatus* obtained from May to August 2013 across peninsular Malaysia. Bins containing height at 50% physical maturity (Ht_m), height at 50% reproductive activity (Ht_r) and height at first capture (Ht_c) are indicated. Ht_m was not included for *H. kelloggi* as all observed individuals were mature.

Chapter 3: When a little equals a lot – a global assessment of seahorses in bycatch

3.1 Introduction

The incidental capture of marine organisms, known as bycatch, generates some of the most significant challenges facing fisheries management today (Hall, 1997), as a great deal of marine life bycatch is retained or discarded, generally without monitoring or regulation (Soykan *et al.*, 2008). A decade ago it was estimated that approximately 8% of the global annual marine catch was discarded. However, some gears, such as tropical shallow water shrimp trawl fisheries, were found to discard an average of 56% of their total catch (Kelleher, 2005). Recent years have seen a move towards retention of bycatch, driven by new markets for bycaught organisms to be used as animal feed, fertilizer or fishmeal (Funge-Smith *et al*., 2005; Kelleher, 2005). This shift may appear to make little difference to wild populations, as discarded bycatch have a low chance of survival (Chopin & Arimoto, 1995; Hill & Wassenberg, 2000). Yet these new markets for bycatch have pushed many bottom trawls beyond economic extinction (Lobo *et al.*, 2010) and encouraged them to indiscriminately maximize the biomass they catch (Hall & Mainprize, 2005; Davies *et al*., 2009). The physical and biological impacts of such indiscriminate practices on fish populations and ecosystems are poorly understood (Chuenpagdee *et al*., 2005), despite clear potential for habitat destruction and population declines.

The majority of bycatch research has focused on large, charismatic megafauna such as sea turtles, seabirds and marine mammals (as reviewed by Lewison *et al.*, 2004) despite the fact that most bycaught species are small fishes maturing at lengths under 20 cm and weights under 100 grams (Alverson, 1994; Kelleher, 2005). Relatively few studies have examined the impacts of bycatch on these species (but see Stobutzki *et al*., 2001a; 2001b; Foster & Vincent 2010a; 2010b; 2012; Arana *et al*., 2013), as it is often assumed that their 'opportunistic' life histories (characterized by fast growth, small body size, low age and size at maturity) make them less vulnerable to the impacts of fishing compared to other large marine animals (Dulvy & Reynolds, 2002; Winemiller, 2005). However, there are caveats to this rule, as certain biological or ecological traits – like demersal habitat preferences or rarity – can increase the vulnerability of certain small fish taxa (Roberts & Hawkins, 1999; Stobutzki *et al*., 2001b; Reynolds *et al.*, 2005; Foster *et al.*, 2010b). However the low value of these species means that very little research attention has been devoted to understanding how bycatch is impacting populations.

Here I focus on seahorses (*Hippocampus* spp.), a genus of incidentally captured small fishes that may be of particular concern because of their biology and ecology. Seahorses share traits with other opportunistic small fishes, including short life spans and rapid growth rates (Curtis & Vincent, 2006). They have other life history attributes, however, including monogamy and parental care, that may make them more susceptible to overfishing (Foster & Vincent, 2004). Ecologically, seahorses are demersal fishes, and *in situ* populations are typically patchy (Foster & Vincent, 2004), such traits may also make them more susceptible to overfishing (Roberts & Hawkins, 1999; Reynolds *et al.*, 2005). Previous research has suggested that the bycatch of seahorses has the potential to affect wild populations (Vincent *et al*., 2011b). For example, fishing can displace individuals or disrupt monogamous pair bonds, thereby reducing population growth. Declines in seahorse catches in indiscriminate gear have been reported by fishers in several countries (Baum & Vincent, 2005; Perry *et al*., 2010) although data are generally very sparse. The patchy distribution of seahorses *in situ* makes it difficult to estimate population abundance, and as even catch and landings data are sparse, it becomes challenging to estimate the global magnitude of seahorse bycatch.

In the absence of global estimates of seahorses in bycatch, volumes reported in international trade have necessarily been used as the next-best estimate of total extraction. A large-scale trade for traditional Chinese medicine, ornamental display and souvenirs led to the listing of seahorses on Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 2002. Implementation of this listing in 2004 meant that all 180 CITES member nations ("Parties") were required to monitor the international trade in seahorses through export and re-export records and prove that such trade was not harming wild populations (CITES, 2004). Of all the seahorses in international trade 95% are inferred to come from bycatch (Vincent *et al.*, 2011b), creating an assumption that CITES international trade volumes are a reasonable proxy for global bycatch volumes. However, as CITES does not monitor domestic trade or the personal use of seahorses, it is likely that CITES international trade volumes severely underestimate bycatch volumes. It is critical to explore the relationship between bycatch and international trade volumes, as this problem is unlikely to go away for CITES Parties. Five shark and all manta ray species (Elasmobranchi), largely obtained through bycatch (FAO, 2013), were recently listed on CITES Appendix II (CITES, 2013) and are certain to face similar assumptions once international trade data becomes available.

This chapter synthesizes published and unpublished literature to conduct the first global analysis of seahorse bycatch using three main approaches. First, I extract data derived from fisher interviews, official data and *in situ* monitoring to estimate catch-per-unit-effort (CPUE) and total volumes of bycaught seahorses extracted in different gear types. Estimates of CPUE can be used to infer the selectivity of different gear types at capturing seahorses in bycatch, and how such selectivity translates into scaled-up total volumes. Second, I compare two methods often used to approximate seahorse bycatch – scaled-up total volumes and international trade data – in order to explore their potential for estimating global bycatch. Lastly, I use information reported from fisher interviews to deduce changes in catch, focusing on catch declines and changes in catch mean size, as a way to assess potential for overfishing. Moving forward, this information will establish conservation and research priorities by determining the "worst offending" gear types. This chapter concludes by considering what these findings for seahorses might suggest for other small bycaught fishes, in particular those whose life history and ecology are likely to make them more vulnerable to fishing impacts.

3.2 Materials and methods

3.2.1 Data sources

This paper attempts to synthesize available literature on seahorses in bycatch, using both published and unpublished sources. The majority of these sources collected data between 1998 and 2001, but dates of data collection ranged from 1989 to 2013. I focused in particular on extracting information that was associated with a specific country and a specific gear type.

The information I extracted from the literature was collected by researchers who used four general data collection approaches: (1) *in situ* interviews with trade participants (e.g. fishers, buyers, importers/exporters, retailers) or those with local knowledge of the seahorse trade (e.g. scientific researchers, non-governmental organizations), (2) official data collected by government agencies detailing either the catch or trade of seahorses (see summary in Vincent *et al.*, 2011a), (3) personal communications sent to Project Seahorse scientists (see summary in Vincent *et al.*, 2011a), and (4) *in situ* direct monitoring of catches or landings of bycaught seahorses. Details on data sources and data collection methods are summarized in Appendix A.

My literature review found that 36 countries reported that seahorses were obtained in bycatch, but only 24 of these countries were used in further analysis. The 12 countries that were left out of further analysis only mentioned that bycatch was known to occur, without providing any quantitative values. Reports from 6 of these 12 countries mentioned the gear types that caught seahorses without providing additional information about those catches, and reports from the other 6 countries only mentioned that bycatch was known to occur (Figure 3.1). For the 24 countries that could be used in further analysis, I extracted data on gear-specific seahorse CPUE, total annual volumes retained in bycatch, and changes in catch and mean size over time (these countries are highlighted in black in Figure 3.1).

3.2.2 Information Extracted

3.2.2.1 Gear types responsible for bycatch

Many different gear types were reported to obtain seahorses across the 30 countries for which such information was available. To eliminate differences in terminology for similar gears and make analysis tractable, I grouped gear types into nine broad categories (dredges, falling gears, gill/entangling nets, hook/lines, scoop nets, seine nets, surround nets, traps, and trawl nets) according to FAO guidelines (Nédélec & Prado, 1990).

3.2.2.2 Catch-per-unit-effort and total volumes of seahorses in bycatch

Estimates of catch-per-unit-effort (CPUE) were reported from fisher interviews, from *in situ* monitoring of catches or landings, and from official statistics. All data were synthesized into gear-specific CPUE estimates for 21 countries. Where ranges or multiple estimates of CPUE were provided, I took the median of these values, and reported both the number of values used to calculate the median as well as the total number of fishers or vessels that generated the values (Appendix B1). In order to make comparisons between CPUE estimates between gear types and countries I converted all estimates to a standard "number of individual seahorses per vessel per day," as per day estimates were the most tractable. For studies that reported annual or monthly CPUEs, these values were divided by 365 or 30 days respectively, or by the actual number of days spent fishing each year or each month if such information was reported. Where catch estimates were reported in kilograms, they were converted to number of seahorses using a standard conversion factor of 372 seahorses per kilogram (Foster *et al*., in revision).

Estimates of total annual catch were available for 17 countries, where authors had used the estimated number of vessels in a particular area to scale up CPUE (Appendix B2). These total annual catch estimates were gear-specific for 15 of the 17 countries. For 11 of the 17 countries, total annual bycatch estimates could be compared to international seahorse export volumes as reported to CITES between 2004 and 2011 (UNEP-WCMC, 2012; Foster *et al*., in revision). Raw data are publicly available (UNEP-WCMC, 2012) and details on how the CITES data were extracted for this analysis can be found in Foster *et al*. (in revision).

3.2.2.3 Changes in catch volumes and mean size

I was able to extract information on reported changes in catch volumes of seahorses and the mean size of seahorses in these catches. Only reports that clearly stated the number of fishers reporting increases, decreases and no change in catch volumes and mean sizes were used to avoid potential biases in reporting. The majority of fisher interviews used in this analysis were conducted between 1995-2000, except for Senegal, which was surveyed in 2013. Comments on changes in catch volumes were available for 594 fishers across 18 countries. Where the reported change was a decline in seahorse catches, 179 fishers in six countries estimated the magnitude of this decline. Fishers also reported on changes in the mean size of captured seahorses (155 fishers, four countries). All studies that reported a change in some characteristic of seahorse catch also reported a temporal reference for that change (i.e. that the change had occurred over the past x years). I calculated the mean temporal reference for each country for which a change in the characteristics of seahorse bycatch had been reported.

3.2.3 Data analyses

Comparison among gear type catch-per-unit-effort (CPUE) and total volumes of seahorses in bycatch were explored to detect differences in gear selectivity and to assess what gear types removed the largest total volumes annually. I used a Kruskal-Wallis rank sum test to compare differences in CPUEs among gears, and used a post-hoc multiple comparison test to examine specific differences between gear CPUEs. Only gear types that had CPUE estimates from three or more countries were included, to minimize biases in the analysis.

In order to examine long-term trends in seahorse bycatch, I used a χ^2 goodness-of-fit test to determine whether fisher responses suggested that (a) catch volumes had increased, decreased, or remained the same over time and (b) the mean size of seahorses in catches had decreased, increased or remained the same over time. First, I tested each individual country, but excluded countries where sample sizes were too low (fewer than 5 fishers had provided responses for each option). Second, I pooled all countries to examine overall trends in seahorse catch volume and mean size.

Where fishers had reported a decline in seahorse catches, researchers asked them to estimate the magnitude of this decline as a percentage over a given time period. I standardized this magnitude as an annual percent decline to adjust for variable time periods. For countries with multiple estimates for magnitude of decline the average was taken, and the overall average was taken across countries. All statistical analyses were completed using the R statistical platform (R Development Core Team 2013). Tests were considered significant when P-values were <0.05.

3.3 Results

3.3.1 Gear types

Thirty countries provided details about which gear types obtained seahorses in bycatch, and these gears were consolidated into nine categories (in accordance with Nédélec & Prado, 1990; Table 3.1). Nearly all countries that provided details about which gear types obtained seahorses in bycatch mentioned trawl nets (29 of 30 countries), and fewer countries reported that gill/entangling nets (16 of 30 countries) and seine nets (14 of 30 countries) caught seahorses in bycatch. However, none of the 30 countries provided information on seahorse bycatch across all nine of the gear type categories, and no country reported the absence of seahorses in a particular gear type category, thereby creating a bias in the available data (Figure 3.2). It is unknown if un-reported gear types were not examined, or if they were examined and ignored because they did not catch seahorses (Figure 3.2, light grey bars).

3.3.2 Volumes of seahorses in bycatch

3.3.2.1 Catch-per-unit-effort

There was a significant difference in CPUE among gear types (Kruskal-Wallis $\chi^2 = 41.24$, d.f. = 4, $p < 0.001$), and a multiple comparison test revealed that gill/entangling nets had significantly lower CPUEs compared to trawl, seine, surround and scoop nets (Figure 3.3). However, a bias in the available literature meant that the majority of CPUE estimates were reported for trawl nets, followed by gill/entangling, surround nets, seine nets and scoop nets (Table 3.2), with fewer estimates for other gear types (Appendix B1). Such limitations meant that only five gears were included in the statistical analysis, as dredges, falling gears, hook and lines and traps had fewer than three countries reporting CPUEs.

3.3.2.2 Total annual volumes

Extrapolated total volumes of seahorses in bycatch reported that approximately 11 million seahorses were obtained across 17 countries, with 80% of this total volume reportedly being caught by trawl nets (Table 3.2). As with CPUE, estimates of total volumes extrapolated in various reports were biased towards trawl gears. This meant that total catch volumes for 10 of the 17 countries were calculated exclusively from trawl data (Table 3.2). Furthermore, whereas catches were extrapolated to the entire country for nine countries, estimates for the remaining eight countries were extrapolated for part of the country only (Table 3.2).

3.3.2.3 Catch volumes compared to reported international trade

CITES annual export volumes were less than the total estimated annual bycatch for the majority of the countries where comparison was possible (Table 3.3). Comparison was possible for 11 countries as they had information on both CITES annual export volumes and total scaled-up estimated annual bycatch volumes. For six of the 11 countries, these CITES annual trade volumes were much less, representing <1% of estimated bycatch volumes for seahorses. However for two countries, Thailand and Peru, CITES annual trade volumes were much higher (>200%) than estimated bycatch volumes (Table 3.3).

3.3.3 Changes in the characteristics of seahorse bycatch over time

Across all countries where data was available, when given the option of reporting that seahorse catches had declined, increased or remained the same over time, the vast majority of fishers reported that seahorse catch volumes had declined ($\chi^2 = 314$, d.f. = 2, $p < 0.001$, Table 3.4). Where individual countries could be examined, fishers in 15 of the 18 countries reported a decline more often than an increase or no change in catches (Table 3.4). Three countries (Bangladesh, Indonesia and Tanzania) showed no significant difference across fisher responses. For those that estimated a magnitude of decline in catch volumes, the standardized annual mean decline across all countries was reportedly ~ 6% $year^{-1}$ with Vietnam reporting the highest annual rate of decline (~13% $year^{-1}$) and Malaysia and Mexico reporting the lowest (~4% $year^{-1}$ respectively; Table 3.4).

Given the option of reporting that the mean size of seahorses in catches had decreased, increased or remained the same over a variable time period, the vast majority of fishers reported that they had observed no change in the mean size of seahorses ($\chi^2 = 120$, d.f. = 2, $p < 0.001$, Table 3.5), although data that met minimum requirements were only available for four countries. Where individual countries could be examined, fishers in all countries reported that they have observed no change in mean size more often than they reported noticing an increase or decrease in mean size.

3.4 Discussion

This chapter confirms that large-scale extraction of seahorses is occurring around the globe across a variety of gear types, despite low CPUEs for each gear type. However, passive selective gears reported lower CPUEs than active indiscriminate gears like trawl nets, which appeared to be extracting the highest volumes of seahorses annually. This chapter also confirmed that international trade data are poor proxies of total extraction in bycatch when compared to scaled-up cumulative annual estimates of seahorses in bycatch. The scaled-up cumulative bycatch estimate across all countries examined in this study was vast, yet each vessel was found to capture very few seahorses in a day. This large global catch appeared to have notable effects on seahorse catches over time according to fishers. One limitation is that the majority of the data used in this study was gathered over a decade ago – and indiscriminate capture has only increased since that time. The scaling-up effect demonstrated here, using seahorses in bycatch, has implications for other small bycaught fishes, especially those that are rare or demersal.

My results suggest that tens of millions of seahorses appear to be extracted annually in bycatch, despite very low CPUE values, likely due to the intensity of fishing effort worldwide. The reported catch-per-unit-effort for seahorse bycatch was extremely low across all gear types (<1–4 seahorse*vessel^{-1}*day^{-1}). These low CPUE estimates were somewhat expected given that seahorses are typically low-density and patchy in their distribution (Foster & Vincent, 2004). Indeed, trade researchers have encountered the argument that such low catch rates preclude indiscriminate gear becoming a problem for seahorses (A. Vincent & S. Foster, pers. obs.). Yet, it is easy to see how the scaling up effect due to the sheer global fishing effort means even low catch rates could indeed pose huge pressures on these small fishes. However, these huge pressures will only be a problem if density compensation is too weak to counteract fishing mortality, or if fishing gear disrupts food web or recruitment dynamics (Winemiller, 2005). For seahorses, high reproductive investment suggests that density compensation is likely low (Foster & Vincent, 2004), and indiscriminate gears impact and alter bottom habitats where seahorses are found (Freese *et al*., 1999).

Active indiscriminate gears showed similar gear selectivity for catching seahorses, while passive gears (gill/entangling nets) showed lower gear selectivity by comparison. The relatively sedentary and demersal nature of seahorses (Foster & Vincent, 2004; Vincent *et al.*, 2005; Caldwell, 2012) suggests that they are less likely to be retained in passive gears that require the fish to encounter the net (i.e. tidal shrimp gill/entangling nets) compared to active gears where the net encounters the fish (i.e. shrimp trawl nets). Indeed, the passive nature of gill/entangling nets means that they are less likely to capture demersal fish in general because they do not drag along the bottom (Nédélec & Prado, 1990). Similar gear selectivity among a range of active indiscriminate gears suggests that these gears may all contribute to seahorse bycatch, even though most research has been focused on bottom trawls.

Extrapolated total volumes showed that trawl gears obtained the greatest number of seahorses in bycatch. Trawl selectivity was, however, no higher than that of other active indiscriminate gears. It is therefore unclear whether this high volume was due to the vast scale of trawl fisheries worldwide or a bias in sampling among gears. The trawl total volumes found in this study supported previous deductions that trawl gears are responsible for the vast majority of seahorse bycatch (Vincent *et al*., 2011b). Yet the results also showed that the field researchers in our source studies biased their sampling and investigations strongly towards trawls. Often this bias was clearly intentional, as researchers frequently reported consulting with local fishers and finding that seahorses were mostly obtained as bycatch in bottom trawls (i.e. Salin *et al*., 2005; Baum & Vincent, 2005; Perry *et al*., 2010). However, similar selectivity across indiscriminate gears indicates that the total volumes reported here, which were biased towards trawls, likely severely underestimate the total extraction of seahorses in bycatch. Indeed, I encourage focused research on other gear types around the world – in particular seine nets, which had a median CPUE for seahorses that was four times greater than trawl nets.

My results show that international trade data reported to CITES are a poor proxy for total extraction of seahorses in bycatch, assuming that the listing of seahorses on CITES Appendix II in 2004 has not triggered a significant reduction in seahorse bycatch. For nearly all countries where data were available, the deduced annual bycatch volumes greatly exceeded the international trade volumes reported to CITES. The only exceptions were Peru and Thailand, which reported export volumes that greatly exceeded the number of seahorses obtained in bycatch each year. Peru's lower bycatch estimates can be explained by sampling limitations, as this study extrapolated bycatch estimates to one small bay only (Alfaro-Shigueto & Mangel, 2011). On the other hand, Thailand does indeed appear to be exporting volumes in excess of what field surveys found in bycatch. This discrepancy is thought to arise from a combination of underreporting by Thai fishers (T-C. Kuo *et al.*, in prep) and imports from neighboring countries (J. Lawson, Project Seahorse, pers. obs.). While the studies used to estimate total extraction of seahorses in bycatch took place prior to the CITES Appendix II listing came into effect in 2004, it is unlikely that the listing caused a significant reduction in seahorse bycatch. Despite temporal discrepancies, this comparison is highly relevant for newly listed CITES Appendix II bycaught marine fishes (ie. sharks and rays: Mundy-Taylor & Crook, 2013): those listings will come into effect in September 2014.

The majority of fishers across all countries reported declines in seahorse catches, which may suggest declines in seahorse populations globally. Such declines have been found for most low-value, small, bycaught fishes throughout Asia (Funge-Smith *et al.*, 2012) and for catches in general around the globe (Watson *et al*., 2013) despite increasing effort. In the absence of time series data to assess fish stocks there is growing support for narrative accounts (Ainsworth *et al.*, 2008). However, using fisher interviews to infer changes in catch must be interpreted with caution (O'Donnell *et al*. 2012). For example, a common indicator of overfishing is a reduction in mean size for a given species over time (Hutchings & Rowe, 2008), yet most fishers in our study reported no size reduction for bycaught seahorses. While this may be true, it could also be because many fishers caught a range of seahorse species, and were unaware of how to tell them apart. I therefore suggest that fisher interviews may not suffice to document certain biological changes, and field sampling remains vital for examining changes in seahorse mean size in response to fishing (Yasué *et al*., in review).

Moving forward with imperfect data is the only way to set research agendas and conservation priorities for bycaught small fishes, despite data limitations and inherent biases in available data. My synthesis draws mostly on studies conducted between 1998 and 2001, but uses information dating from 1989 to 2013. For CPUE, I tacitly ignore temporal change across decades, despite likely increases in fishing effort and associated declines in CPUE (Watson *et al*., 2013). Additionally, I acknowledge a strong bias towards bottom trawl vessels in this chapter, yet this bias was by no means a coincidence – these fishing fleets are the most destructive and have become increasingly indiscriminate in their catches (Funge-Smith *et al*., 2005; Kelleher, 2005), and authors often explicitly focused on these gears because they were found to retain large numbers of seahorses in bycatch. Seahorses were reportedly absent in the bycatch of offshore trawl fisheries in Peru, which suggests that seahorses may be absent from deepwater trawl bycatch (48-64 km offshore, 180-250 m depth; Vincent *et al.*, 2011a), but data limitations prevented any further investigations into how seahorse bycatch varied with water depth. Under any management regime, it can be challenging to tackle bycatch issues. Where capacity and resources are very limited, as is the case for many of the countries where seahorses are obtained in bycatch, the challenges become enormous, and it is critical to move forward with available – albeit imperfect – data.

If such small CPUE for seahorses in bycatch adds up to many millions of animals, the implications are potentially alarming for unmonitored bycaught species that are retained at higher CPUEs, especially for those that are rare or have patchy distributions. Small species attract little or no attention for their economic or ecological value, and therefore are often lumped together (Funge-Smith *et al.*, 2012) and are unmonitored and unregulated (Soykan *et al.*, 2008). Yet many are caught at much greater catch-per-unit-effort rates than seahorses, leading to large cumulative tallies. In a northern Australia shrimp trawl fishery, small fishes were found to make up the majority of the teleost bycatch (both by weight and by number of individuals) and the vast assortment of species retained in bycatch meant that most species was encountered rarely (Stobutzki *et al*., 2001a). The three species of small fishes that had the highest CPUE were caught at rates ranging from 288.01 – 380.87 individuals*trawl^{-1}*hour^{-1} (Stobutzki *et al*., 2001a), meaning that for every <1–4 seahorses, vessel^{-1} day^{-1}, hundreds to thousands of other small fish species are being caught as bycatch in many countries around the world with little to no monitoring. While some small fish species may be able to withstand such intense fishing pressure, many others may not, and the impacts of the large-scale removal of these small fishes on food web and ecosystem functioning – not to mention the implications for human food security – have not been considered for most of these species (Pauly *et al*., 2005; Tacon & Metian, 2009; Funge-Smith *et al.*, 2012).

The urgency in managing and understanding small fishes in bycatch is growing now that many trawl fisheries are becoming directed at maximizing fish biomass for fish meal and fertilizer, with no target species (Funge-Smith *et al.*, 2005). Any economic checks have essentially disappeared (Lobo *et al.*, 2010), in part because of fuel subsidies (Sumaila *et al.*, 2010). This means that gear modifications or programs to reduce pressure on wild populations of non-target small species are unlikely to be supported by fishers. This is unfortunate as the generation of incentives for fishers to avoid bycatch has driven some of the most successful bycatch reduction programs (Branch & Hilborn, 2008). Perhaps the most promising catalyst for a reduction in destructive and unsustainable fishing practices is pressure from the general public on government and industry. Seahorses are among the best marine animals to draw attention to the wasteful practices associated with bottom trawling for the plethora of marine life that are retained as bycatch and are largely ignored.

Table 3.1 Gear types reported to catch seahorses by country (n=30 countries) and the corresponding FAO gear type category.

Reported Gear Type	FAO Gear Type Category	Countries Reporting
Oyster/other Molluscs	Dredges	Malaysia, Mexico
Cast nets	Falling gear	Brazil, India, Kenya, Panama, Senegal, Tanzania, Thailand
Drift nets	Gill/entangling net	India, Malaysia, Senegal
Tidal shrimp nets	Gill/entangling net	Indonesia, Malaysia, Senegal
Trammel nets	Gill/entangling net	Bangladesh, India, Malaysia, Portugal, Senegal
Gill nets	Gill/entangling net	Bangladesh, Costa Rica, Ecuador, Guatemala, Honduras, India, Indonesia, Malaysia, Mexico, New Zealand, Nicaragua, Pakistan, Panama, Peru, Philippines, Portugal, Tanzania
Tangle nets	Gill/entangling net	Senegal
Longline	Hook and line	Malaysia, Senegal
Bag nets	Scoop net	Bangladesh
Push nets	Scoop net	Philippines, Portugal, Thailand
Larval nets	Scoop net	Bangladesh
Beach seine	Seine net	Bangladesh, Brasil, Costa Rica, Ecuador, Honduras, India, Indonesia, Kenya, Mexico, Mozambique, Philippines, Portugal, Senegal, Tanzania
Shore seine	Seine net	India
Dragnets	Seine net	India

Reported Gear Type	**FAO Gear Type Category**	**Countries Reporting**
Purse seine	Surround net	Ecuador, India, Indonesia, Kenya, Malaysia, Peru, Senegal, South Korea, Tanzania, Thailand
Surround net	Surround net	Tanzania
Fish traps	Trap	Indonesia, New Zealand
Crab traps	Trap	India, Indonesia, New Zealand
Basket traps	Trap	Tanzania
Enclosure pens	Trap	Philippines
Trap nets	Trap	India, Tanzania
Trawl	Trawl net	Australia, Bangladesh, Belize, Brasil, China, Costa Rica, Ecuador, France, Guatemala, Honduras, India, Indonesia, Kenya, Malaysia, Mexico, Mozambique, New Zealand, Nicaragua, Nigeria, Pakistan, Panama, Peru, Philippines, Portugal, Senegal, Tanzania, Thailand, United States, Vietnam
Seahorse bycatch was mentioned to occur, the responsible gear type was not reported		Argentina, Croatia, Spain, Taiwan, Turkey

Table 3.2 Median (M) catch-per-unit-effort (seahorses^{-1} *vessel^{-1}*day^{-1}) for five categories of fishing gear. Median values were calculated from estimates provided by fishers during trade interviews, from official data sources or from *in situ* monitoring carried out in 21 countries. Associated with each median CPUE is the range of CPUE estimates that were provided by fishers during interviews. Also included, where data was available, is the estimated annual total catch (in number of seahorses) for trawl nets, for other gear types excluding trawl nets and for all gear types (including countries where gear type was not specified), as extrapolated by the authors of each report. The area that the author extrapolated the total catch over is also listed, only where total catch volumes were extrapolated. Countries are listed alphabetically.

Country	Catch-per-unit-effort										Total catch		
	Trawl nets		*Gill/entangling nets*		*Surround nets*		*Seine nets*		*Scoop nets*		Trawl nets	All other gears	Total catch
	M	Range	M	Range	M	Range	M	Range	M	Range			
Australia[1]	0.01												
Bangladesh[3] Chittagong	0.01	0.011-0.274	0.65	0-1.97					0.92	0.04-2.53		1,221,937	1,221,937
Belize[1] Dangria to Plascencia	11.50	3.0-20.0									18,402		18,402
Costa Rica[1]	4.76												
Ecuador[2] Gulf of Guayaquil	0.69	0.5-1.0									51,000		51,000
Guatemala[1] ~whole country	0.37	0.03-1.07	0.02	0.01-0.03							37,960		37,960
Honduras[1]	7.23	0-50	0.34	0.01-							267,288		267,288

Country	Catch-per-unit-effort										Total catch		
	Trawl nets		*Gill/entangling nets*		*Surround nets*		*Seine nets*		*Scoop nets*		Trawl nets	All other gears	Total catch
	M	Range	M	Range	M	Range	M	Range	M	Range			
Roatan Island				0.67									
India[4,8,9] ~whole country	1.00	0-326.0			1.5	0.14-2.86	5.18	0-42.86			3,209,597		3,209,597
Indonesia[4]			0.52		4.15	0.14-20.0	5.71	2.0-9.0					
Malaysia[5,11,12] ~whole country	0.55	0.01-2.64	0.79	0-10.0	0.02						645,233		645,233
Mexico[2] ~whole country	0.33	0-1.97									330,864		330,864
Nicaragua[1] ~whole country	6.17	1.5-10.0									187,548		187,548
Pakistan[4] Karachi	0.33	0-3.29	1.29	0-3.47									83,400

Country	Catch-per-unit-effort										Total catch		
	Trawl nets		*Gill/entangling nets*		*Surround nets*		*Seine nets*		*Scoop nets*		**Trawl nets**	**All other gears**	**Total catch**
	M	**Range**	**M**	**Range**	**M**	**Range**	**M**	**Range**	**M**	**Range**			
Panama[1] Pacific Coast	0.24	0-1.0									20,260		20,260
Peru[2,13] Sechura Bay	0.86	0.29-2.86	2.36	1-1.72	14.29	6.0-20.0						25,000	25,000
Philippines[6] ~whole country	3.00	2.0-4.0					2	1.0-3.0	2	1.0-3.0	553,350	239,925	793,275
Senegal[14] ~whole country	23.22	13.56-32.88	0.04	0-1.47	0	0-1.47	0.21	0-0.88			114,023	257,544	371,567
Tanzania[1,18] ~whole country	0.42	0.33-0.50			1	0.03-100							42,000
Thailand[5,15,16] ~whole country	1.14	0-2.88	0.07		1.5	1.33-1.67			31.48	0.49-62.5	1,170,638	251,160	1,421,798
USA[1,10]	0.03	<0.01-											

Country	Catch-per-unit-effort										Total catch		
	Trawl nets		*Gill/entangling nets*		*Surround nets*		*Seine nets*		*Scoop nets*		Trawl nets	All other gears	Total catch
	M	Range	M	Range	M	Range	M	Range	M	Range			
Vietnam[7,17] South-central coast	1.55	0.55 0.25-2.5	0.04	0-1.47							2,067,625		2,067,625
Overall	**0.80**	**0-326**	**0.52**	**0-10**	**1.50**	**0-100**	**3.59**	**0-42.86**	**2**	**0.04-62.47**	**8,673,788**	**1,995,566**	**10,794,754**

Sources: [1] Vincent *et al.*, 2011a; [2] Baum & Vincent, 2005; [3] B. Giles, Project Seahorse, unpublished data; [4] A. Perry, Project Seahorse, unpublished data; [5] Perry *et al.*, 2010; [6] M. Pajaro, Project Seahorse, unpublished data; [7] Giles *et al.*, 2006; [8] Salin *et al.*, 2005; [9] Murugan *et al.*, 2011; [10] Baum *et al.*, 2003; [11] J. Lawson, Project Seahorse, personal observation; [12] Choo & Liew, 2005; [13] Alfaro-Shigueto & Mangel, 2011; [14] A. Cisneros-Montemayor, Project Seahorse, unpublished data; [15] T-C. Kuo, Project Seahorse, unpublished data; [16] Laksanawimol *et al.*, 2013; [17] Meeuwig, *et al.*, 2006; [18] McPherson & Vincent, 2004

Table 3.3 Comparing the ratio of annual total bycatch estimates (this study, years indicated) to annual average seahorse exports as reported to CITES across 2004-2011 (UNEP-WCMC, 2012; Foster *et al*., in revision); both volumes are in total number of individual seahorses. Only countries for which both estimates were available are included in this table. The region for which total annual bycatch was estimated from is also included.

Country	Estimated total annual bycatch	CITES average annual exports (2004-2011)	Ratio of exports to bycatch	Year(s) surveyed for bycatch	Region for which bycatch was estimated
Belize	18,402	63	0.34	2000	Dangria to Plascencia
Ecuador	51,000	19	0.04	2000	~whole country
Malaysia	919,747	107,019	11	1998-1999	~whole country
Mexico	330,864	40,284	12	2000	~whole country
New Zealand	1,115	2	0.18	1989-2001	~whole country
Panama	20,260	1	<0.01	2000	~Pacific coast
Peru	25,000	64,993	260	2000	Sechura Bay
Philippines	793,275	2,650	0.33	1998-2001	~whole country
Senegal	371,567	234,852	63	2013	~whole country
Thailand	2,447,586	4,970,420	203	1998-2013	~whole country
Vietnam	2,116,875	9,280	0.2	1995-2000	South-central coast

Table 3.4 The total number of fishers (n), and those reporting an increase, decrease or no change (stable) in bycatch volumes of seahorses over time, as well as the mean time period over which these estimates were made (from the year that the study was conducted, also shown). The mean annual magnitude (%) and the mean time period for over which that magnitude was standardized are included. * indicates χ^2 goodness-of-fit test p-value >0.05 for that individual country. Countries are listed alphabetically.

Country	Year	*Change in catch volume*					*Magnitude of decline*		
		Total (n)	Increase (n)	Decrease (n)	Stable (n)	Time Period	Total (n)	Annual Magnitude (% ± S.D.)	Time Period
*Bangladesh[3]	1999	23	7	8	9	11			
Belize, Costa Rica, Nicaragua, Panama, Peru[2]									
	2000	36	0	21	15	19			
Ecuador[2]	2000	15	0	13	2	19			
Guatemala[2]	2000	7	0	5	2	19			
Honduras[2]	2000	12	0	10	2	19			
India[4]	1999	160	23	80	57	10	17	5.8 (2.2)	10
*Indonesia[4]	1990-2002	35	9	16	10	10			
Malaysia[5]	1998-1999	52	1	37	14	17	25	4.1 (1.5)	17

Country	Year	*Change in catch volume*					*Magnitude of decline*		
		Total (n)	Increase (n)	Decrease (n)	Stable (n)	Time Period	Total (n)	Annual Magnitude (% ± S.D.)	Time Period
Mexico[2]	2000	45	0	39	6	19	23	4.1 (0.5)	22
Pakistan[6]	1999	11	0	8	3	11	6	4.7 (N/S)	11
Senegal[8]	2013	4	0	4	0	20			
*Tanzania[1]	2000	14	8	2	4	15			
Thailand[5]	1998-1999	37	0	30	7	9	19	6.7 (6.0)	9
Vietnam[7]	1995-1999	143	7	122	14	4	89	12.8 (6.1)	4
Overall		594	55	395	145	14	179	6.4 (3.3)	12

Sources: [1] Vincent *et al*., 2011a; [2] Baum & Vincent, 2005; [3] B. Giles, Project Seahorse, unpublished data; [4] A. Perry, Project Seahorse, unpublished data; [5] Perry *et al*., 2010; [6] Project Seahorse, unpublished data; [7] Giles *et al*., 2006; [8] A. Cisneros-Montemayor, Project Seahorse, unpublished data; [9] McPherson & Vincent, 2004

Table 3.5 The number of fishers reporting an increase, decrease or no change in size of bycaught seahorses over time, and the mean time period for which that change was reported (from the year that the study was conducted, also shown). Countries are listed alphabetically.

Country	Year	Increase	Decrease	No Change	Time Period
India[4]	1999	1	7	58	10
Malaysia[5]	1998-1999	0	12	26	19
Senegal[3]	2013	0	11	5	21
Thailand[5]	1998-1999	0	12	24	17
Overall		1	41	113	15

Sources: [1] A. Perry, Project Seahorse, unpublished data; [2] Perry *et al*., 2010; [3]A. Cisneros-Montemayor, Project Seahorse, unpublished data

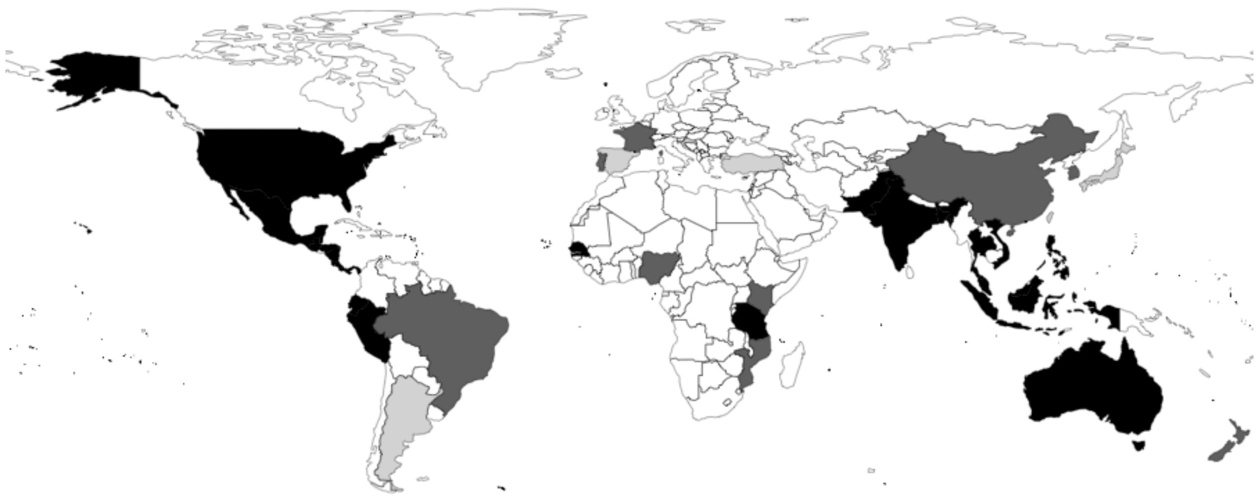


Figure 3.1 The distribution of countries included in my analysis of seahorse bycatch. Light grey countries (n = 6) provided only evidence (through personal communication) for the bycatch of seahorses (did not gather information on bycatch), dark grey countries (n = 6) only mentioned bycatch to occur for a particular gear type but did not gather information on seahorse bycatch for this gear type, while black countries (n = 24) are those for which I was able to extract information on volumes, price paid to fishers and changes over time (where data was available) for seahorses in bycatch. Countries without shading did not contribute data to my analysis.

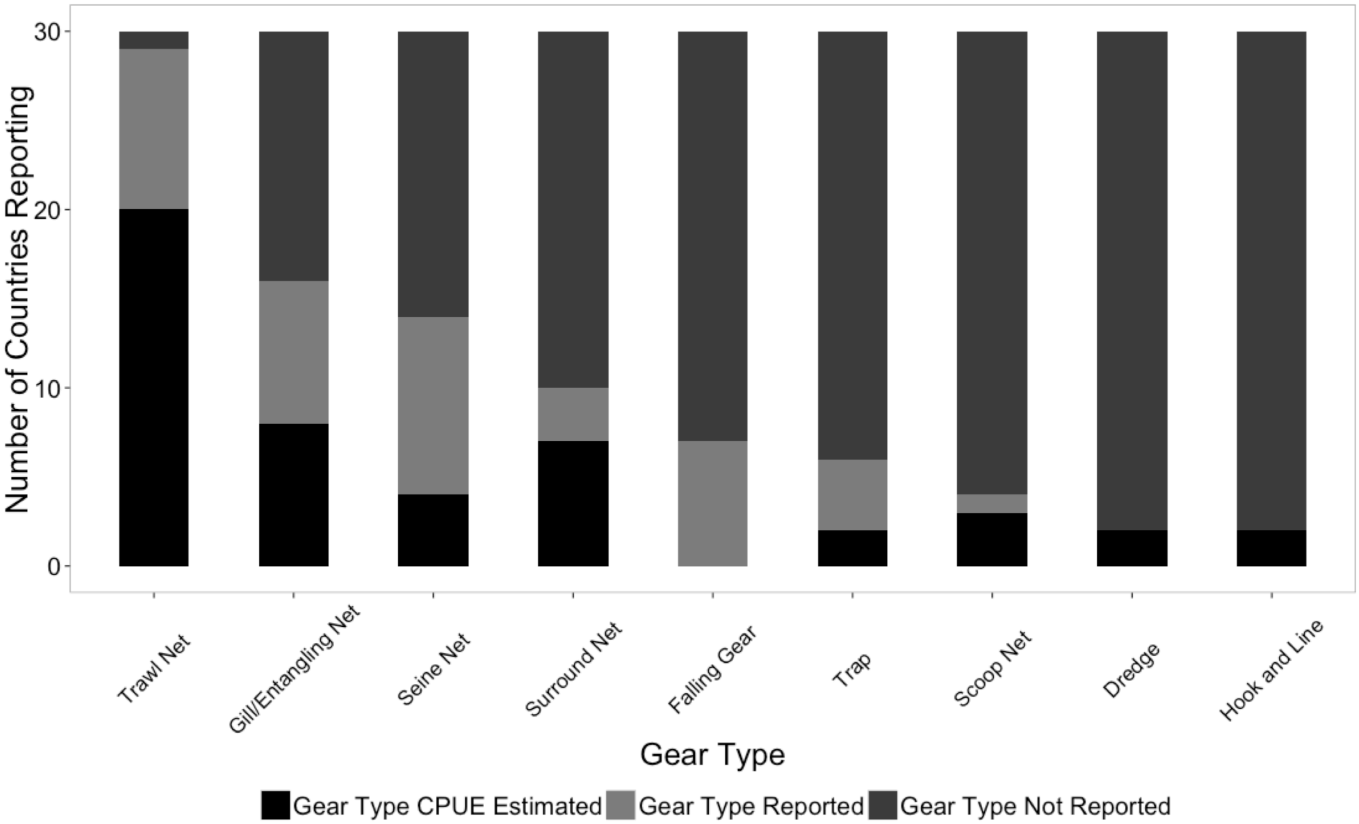


Figure 3.2 The frequency by which a gear type was reported to incidentally capture seahorses across countries. Gear types are categorized according to FAO guidelines (Nédélec & Prado, 1990). Black bars indicate number of countries for which a gear-specific CPUE was available. Dark grey bars indicate number of countries for which a gear type was reported to catch seahorses but no CPUE information was available. Light grey bars indicate countries for which the gear type was not mentioned in the report, but it is unknown if that gear type was investigated and found not to catch seahorses, or was not investigated at all.

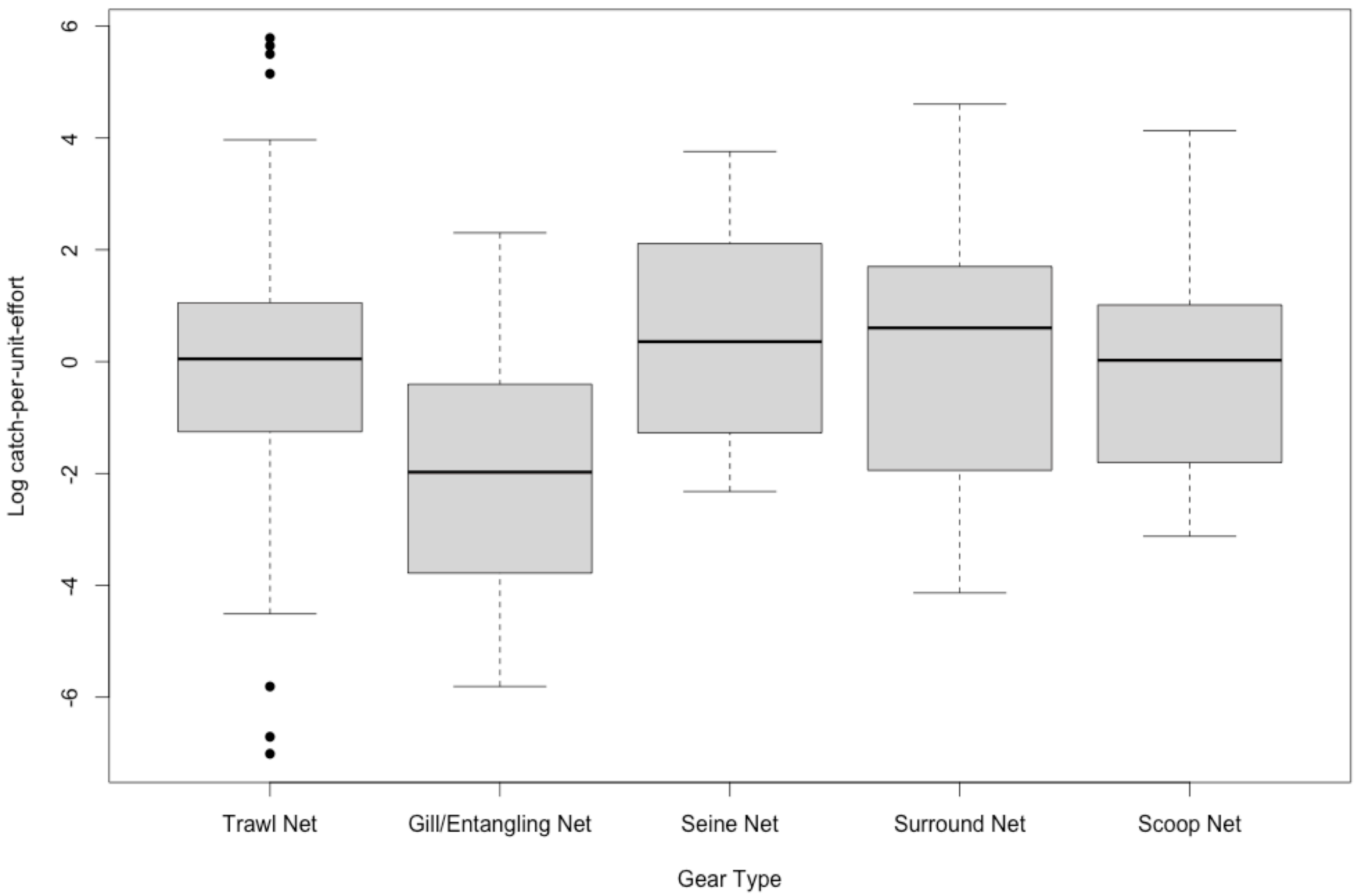


Figure 3.3 Box plots showing log-transformed catch-per-unit-effort (CPUE; seahorses^{-1} *vessel^{-1}*day^{-1}) across five different gear types. Catch-per-unit-effort was calculated from trade reports for 21 countries from estimates provided by fishers during interviews, official data and *in situ* surveys. Box plot whiskers represent the interquartile ranges of the log-transformed CPUEs.

Chapter 4: Conclusion

The trends and practices associated with indiscriminate and nonselective fishing may have significant impact on fish populations and the habitats that sustain them, and the scientific community is in need of a way to communicate these concerns. Seahorses are one such communication tool, as they are a charismatic genus that can act as a flagship group of species for small, bycaught fishes that make up the majority of non-target catch around the globe.

My thesis assessed the impacts of bycatch on seahorses on both a national and global scale. Nationally, data-poor methods captured essential life history information and identified differences among species and fished populations within peninsular Malaysia. On a global scale, my thesis identified gear types responsible for the highest CPUEs and volumes of seahorse bycatch. Such data-poor methods may be applied to other bycaught small fishes, as so little is known about many of these species. My global assessment encourages more research on small fishes in bycatch, using seahorses as a flagship taxon.

4.1 Assessing national fishing impacts

My research generated critical life history information for three bycaught species of seahorse for which biological data are limited. Such information included sex ratios, height-mass relationships and heights at maturity. My research was the first to report life history parameters for *H. kelloggi*, and provided new information for *H. spinosissimus* and *H. trimaculatus*. Some of the life history parameters that were generated were rapidly applied to assess fishing impact for these three species while others have laid the groundwork for future research.

I used two estimates of height at maturity, a key life history parameter, to rapidly assess three bycaught seahorse species of concern. Height at maturity is commonly used in data poor situations, but for seahorses it had been inconsistently defined in previous life history studies (as noted in Foster & Vincent, 2004). Therefore I used two estimates of maturity to assess impacts: height at physical maturity (males with a visible but empty and taut brood pouch) and height at reproductive activity (males with an active brood pouch). My results showed that applying data-poor assessment methods using these two definitions produced strikingly different results. For example, when I pooled catches across the Malay Peninsula to examine the percentage that was over height at maturity (in accordance with Froese, 2004) over half of the *H. trimaculatus* catch were physically mature, while only 5% were reproductively active. These results raised concern that *H. trimaculatus* may perhaps be of greater conservation concern than the other two species in peninsular Malaysia because they were being caught before they were able to contribute to the next generation.

I used fisheries-dependent port surveys to assess catches of *H. trimaculatus* in different regions of peninsular Malaysia to look for signs of overfishing. I was able to identify a fished population of *H. trimaculatus* in the southwest region of peninsular Malaysia that showed indicators of overfishing. This population had a heavily skewed female-bias sex ratio in catches (only 6% males), and a notably smaller mean height in sampled catches than *H. trimaculatus* in other regions of the peninsula. In the absence of long term monitoring, this information cannot confirm that overfishing is occurring, but can at least raise concern and draw attention to worrying trends in the southwest region of the peninsula.

In summary, Chapter 2 used data-poor techniques to rapidly prioritize both species and fished populations for conservation concern. In order to determine whether these three species are overfished, future efforts should focus on estimating abundance and fishing mortality (Walters & Martell, 2004) or gathering fisheries-independent data on natural mortality and growth rates (Hutchings, 2003). Nonetheless, the data-poor rapid assessment techniques described in this thesis are useful for comparative studies among species (Dulvy & Reynolds, 2002) or for examining catches within countries. Moreover, setting aside problems with limited data, the reality is that we have to start taking action on the basis of rapidly obtained data. We will probably never have estimates of biomass or even time series data for these fisheries – and certainly cannot wait for such data given the urgency for conservation action.

4.2 Assessing global fishing impacts

In Malaysia, fisheries-dependent port surveys identified areas where conservation concern should be focused, so I decided to use a similar approach to examine the gear types that obtained seahorses in bycatch on a global scale. However, available literature on seahorses in bycatch used a range of different approaches, meaning that fisheries-dependent port surveys were necessarily combined with data from fisher interviews and formal data records to describe trends in seahorse catches. While fisher interviews can sometimes exaggerate catch rates (O'Donnell, 2012), they also provide valuable information in the face of limited data (Johannes, 1998).

Whatever the accuracy of fisher accounts of bycatch, the number of seahorses obtained each day for each fisher was low across all gear types, yet these catch rates scaled up to tens of millions of seahorses being extracted each year in bycatch. However, passive gears like gill/entangling nets showed greater selectivity, and caught fewer seahorses for a given unit of effort compared to indiscriminate gears like bottom trawls. Researchers biased sampling efforts towards trawl gears, and as a result the greatest volume of seahorses in bycatch were reportedly sourced from these gears. However, because all indiscriminate gears had similar CPUEs, this number likely underestimates total annual extraction of seahorses in bycatch.

Fisher interviews played a key role in gathering information on the global picture of seahorses in bycatch, to varying degrees of usefulness and reliability. It was surprising that many fishers were able to assign a CPUE to seahorses, or notice trends in catches, given a tendency by fishers to ignore bycatch unless penalties are developed to encourage avoidance (Branch & Hilborn, 2008). Yet the cultural and medicinal value, small financial incentive and unique body shape of seahorses seems to make them more notable to fishers (J. Lawson, pers. obs.). Globally, fishers overwhelmingly felt that seahorse catches had declined over time, raising concern about the sustainability of seahorses in bycatch, yet they did not report evidence of a decline in mean size, which is inferred to be occurring for *Hippocampus trimaculatus* in Chapter 2. The latter result was not surprising, considering that most fishers obtained a mixture of multiple seahorse species, and did not distinguish between them.

Previous studies have looked at the characteristics of seahorses in bycatch in isolation (Baum *et al*., 2003; McPherson & Vincent, 2004; Baum & Vincent, 2005; Giles *et al*., 2006; Perry *et al*., 2010; Vincent *et al*., 2011a) and my study was the first to examine global trends. Prior to my synthesis of global data, data limitations meant that it was often inferred that global bycatch numbers could be a reasonable proxy for global trade numbers, as bycatch is inferred to provide about 95% of the seahorses in export (Vincent *et al.*, 2011b). My research reveals that total annual bycatch volumes in most countries were much higher than those reported in the CITES database (UNEP-WCMC, 2012). Presumably this is because many seahorses are entering unregulated domestic trade, entering international trade illegally or being discarded – meaning that they were not officially monitored by CITES. This analysis of seahorses in bycatch provides a cautionary tale for other bycaught species on CITES Appendix II, such as the recently listed sharks and rays (CITES, 2013). As some of the first marine fishes added to CITES Appendix II since its inception, seahorses provide a rich opportunity to examine the challenges and opportunities that arise from international trade data.

Seahorses can serve as flagship species for the vast assortment of fish taxa that are retained as bycatch, and are largely unmonitored and ignored. No matter how scarce the data is on seahorses, it is still often better than the data on other small marine fishes obtained in bycatch. The decision to list seahorses on CITES Appendix II was facilitated by access to a body of research devoted to understanding the biology and trade of seahorses (summarized in Foster & Vincent, 2004; Vincent *et al.*, 2011b). This relative wealth of data makes seahorses unique among small bycaught fishes. Life history parameters have been described for several species of seahorse (Curtis & Vincent, 2006; Morgan & Vincent, 2013), allowing for empirical analyses to fill gaps for lesser known species. Trade surveys have characterized seahorses in bycatch by using port surveys and fisher interviews. Fishers are often able to recall catch rates and changes over time because seahorses have a unique body shape and an important spiritual and medicinal role in many cultures. This relative wealth of information (when compared to other small fishes) makes seahorses ideally suited to draw attention to bycatch issues.

4.3 Application of data-poor methods for small fishes

While there is a relative wealth of data for seahorses when compared to other small bycaught fish species, the reality is that available data still pale in comparison to data-rich fish stocks. As a result, I applied and modified common and novel approaches to examine bycatch fishing impacts on seahorses. Some of these techniques may be possible for some bycaught fishes. For many, however, even the simple assessment techniques applied here will be impossible. This raises the question – how do we provide solutions for small bycaught fish species where even the most basic catch or life history data is limited or entirely unknown?

In the face of limited data we desperately need ways to assess impacts for small fishes, and in some cases broad brushstroke comparisons can be used to prioritize species for conservation (Dulvy and Reynolds, 2002). Chapter 2 showed that comparative life history approaches can reveal species, and fished populations within species, that may be more vulnerable to overfishing. Chapter 3 revealed that the "worst offending" gear types and countries could be identified using seahorse bycatch information reported by fishers in combination with *in situ* port surveys and official data. These approaches can be combined to set research priorities for small fishes in bycatch. For example, my research was able to identify countries with notably high catch rates of seahorses (i.e. indiscriminate trawl fishers in Senegal, West Africa). This sets a research agenda for studying the life history of bycaught seahorses found in Senegal's waters. This combination approach allows areas where fishing impacts are greatest and species vulnerability is highest to be prioritized for conservation efforts.

4.4 Implications of small fishes in bycatch

Excessive take of small fishes in bycatch is a problem that truly should raise concern in the scientific community and general public, given their extraction may have major implications not only for the effective functioning of marine food webs, but for global food security as well. Small fishes play such a critical role in food web functioning (Alder *et al*., 2008) and while their removal in bycatch may be poorly understood (Chuenpagdee *et al*., 2005), it is clear that this wholesale removal will have impacts on global food security. The world's poorest countries rely heavily on small fishes not only as a source of income, but also as a vital source of protein (La Manach *et al*., 2012). Serial depletion and fishing down marine food webs means that fishing pressure on small fishes will continue to increase (Pauly *et al*., 2002). Many small fishes are characterized as resilient to fishing pressure because of their opportunistic life history strategy

(Jennings *et al*., 1998; Winemiller, 2005), yet studies have shown that small fish species that make up the majority of the bycatch biomass can still show signs of overfishing (Foster & Vincent, 2010b). Additionally, rare, demersal and patchily distributed small fishes have been shown to have greater vulnerability to fishing pressure (Dulvy *et al*., 2003, Stobutzki *et al*., 2001b).

The trend of maximizing biomass in bottom trawling fisheries may denote the end of target species for this gear type, as serial depletion of target species has opened new markets for bycatch (Funge-Smith *et al*., 2005). The path to serial depletion of small fishes in bycatch has arisen only in the past few decades. Pauly (1996) noted that trawlers in Southeast Asia would often sew mosquito netting overtop their trawl codends in order to maximize catches of targeted shrimps or anchovies, however he made no mention of any attempt to maximize bycatch. Subsequently, a global shift occurred from discarding bycatch to retaining bycatch (Kelleher, 2005). More recently, several studies have noted alarming situations when the value of the bycatch creates an incentive for indiscriminate fishing aimed at maximizing biomass. For example Lobo *et al.* (2010) noted that in response to declines of target catches, fishers in India had started maximizing bycatch, which has driven these fisheries beyond economic extinction. Similar trends of maximizing biomass through small mesh sizes have been found in Indonesia and several African nations (Davies *et al*., 2009). Seldom if ever, however, have fisheries reached the apogee of random take that characterizes bottom trawl fisheries in much of Southeast Asia now (Funge-Smith *et al.*, 2005). Finally, indiscriminate fishing practices not only damage fish populations and habitats, but also encourage troubling economic (Lobo *et al.*, 2010; Sumaila *et al.*, 2010) and social (Sylwester, 2014) practices.

As we look ahead to the future of our oceans, we must recognize the impacts that indiscriminate fishing practices have not only on ecosystems, but on society as well. Trends in the bottom trawl fisheries of Southeast Asia may be a glimpse into the future of severely overfished regions. Not only are fished populations and critical habitats severely damaged, but also they are done so at the expense of poor and marginalized people – both in terms of food security and basic human rights. Scientists need to spend more time and effort on non-selective fisheries in developing countries, always looking for data-limited solutions to advance effective management. Even more important, however, is the needed for a groundswell of public support to put pressure on governments and consumers to eradicate indiscriminate fishing practices and encourage sustainable fisheries.

Seahorses are among the best marine fishes to highlight the impacts of nonselective gear on small fishes. In the 1990s, public outcry in support of marine mammals and sea turtles captured in nonselective gear set the agenda for funding agencies and scientists and resulted in gear modifications and consumer awareness (Hall, 1997; Hall *et al*., 2000). The bycatch problem of the 2000s is undoubtedly bottom trawl fisheries, and one way we can communicate this issue and generate attention is through seahorses, acting as a flagship genus for small bycaught fishes.

In closing, my thesis is contributes to what will hopefully become a seminal shift in fisheries science – evolving from focusing exclusively on target species in developed countries, to tackling issues with non-target species in developing nations. My in-depth investigation into one taxon of small, bycaught fish demonstrates that intrinsic vulnerability (Chapter 2) and magnitude of fishing pressure (Chapters 2 and 3) can be used to assess potential for overfishing. While it impossible to assess the numerous species of small fish caught in bycatch across the globe, my thesis at least uses one genus of small bycaught fish as a flagship for all small bycaught fishes to generate conservation concern.

Bibliography

Aguilar, A. & Lockyer, C. H. (1987). Growth, physical maturity, and mortality of fin whales (*Balaenoptera physalus*) inhabiting the temperate waters of the northeast Atlantic. *Canadian Journal of Zoology* **65**, 253–264.

Ainsworth, C. H., Pitcher, T. J. & Rotinsulu, C. (2008). Evidence of fishery depletions and shifting cognitive baselines in Eastern Indonesia. *Biological Conservation* **141**, 848–859.

Alder, J., Campbell, B., Karpouzi, V., Kaschner, K. & D. Pauly. (2008). Forage fish: from ecosystems to markets. *Annual Reviews in Environment and Resources* **33**, 153–166.

Alfaro-Shigueto, J. & Mangel, J. C. (2011). The seahorses of Sechura Bay, Peru: a flagship species for artisanal fishery sustainability. In: Final Report to Duke University and Oak Foundation Mini-grants in Marine Conservation.

Alverson, D. L. (1994). A global assessment of fisheries bycatch and discards. *Fisheries Technical Paper No 339*. Rome: Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations.

Andrew, N. L. & Pepperell, J. G. (1992). The by-catch of shrimp trawl fisheries. *Oceanography and Marine Biology Annual Review* **30**, 527–565.

Arana, P. M., Wehrtmann, I. S., Orellana, J. C., Nielsen-Muñoz, V. & Villalobos-Rojas, F. (2013). By-catch associated with fisheries of *Heterocarpus vicarius* (Costa Rica) and *Heterocarpus reedi* (Chili) (Decapoda: Pandalidae): A six-year study (2004-2009). *Journal of Crustacean Biology* **33**, 198–209.

Bailey, M., Sumaila, U. R. & Martell, S. J. D. (2013). Can Cooperative Management of Tuna Fisheries in the Western Pacific Solve the Growth Overfishing Problem? *Strategic Behavior and the Environment* **3**, 31–66.

Baum, J. K. & Vincent, A. C. J. (2005). Magnitude and inferred impacts of the seahorse trade in Latin America. *Environmental Conservation* **32**, 305–319.

Baum, J. K., Meeuwig, J. J. & Vincent, A. C. J. (2003). Bycatch of lined seahorses (*Hippocampus erectus*) in a Gulf of Mexico shrimp trawl fishery. *Fishery Bulletin* **101**, 721–731.

Boisseau, J. (1967). *Régulations hormonales de l'incubation chez un vertébré mâle: recherches sur la reproduction de l'Hippocampe*. PhD thesis, Bordeaux, University of Bordeaux. France.

Branch, T. A. & Hilborn, R. (2008). Matching catches to quotas in a multispecies trawl fishery: targeting and avoidance behavior under individual transferable quotas. *Canadian Journal of Fisheries and Aquatic Sciences* **65**, 1435–1446.

Branch, T. A., Jensen, O. P., Ricard, D., Ye, Y. & Hilborn, R. (2011). Contrasting global trends in marine fishery status obtained from catches and from stock assessments. *Conservation Biology* **25**, 777–786.

Cai, N., Xu, Q., Yu, F., Wu, X. & Sun, G. (1984). Studies on the reproduction of the seahorse *Hippocampus trimaculatus*. *Studia Marina Sinica* **23**, 83–92.

Caldwell, I. R. (2012). *Habitat use, movement, and vulnerability of sedentary fishes in a dynamic world*. PhD thesis, Vancouver, The University of British Columbia. British Columbia, Canada.

Carruthers, T. R., Punt, A. E., Walters, C. J., MacCall, A., McAllister, M. K., Dick, E. J. & Cope, J. (2014). Evaluating methods for setting catch limits in data-limited fisheries. *Fisheries Research* **153**, 48–68.

Choi, Y-U., Rho, S., Park, H-S. & Kang, D-H. (2012). Population characteristics of two seahorses, *Hippocampus coronatus* and *Hippocampus mohnikei*, around seagrass beds in the southern coastal waters of Korea. *Ichthyological Research* **59**, 235–241.

Choo, C. K., & Liew, H. C. (2003). Spatial distribution, substrate assemblages and size composition of sea horses (Family Syngnathidae) in the coastal waters of Peninsular Malaysia. *Journal of the Marine Biological Association of the United Kingdom* **83**, 271–276.

Choo, C. K., & Liew, H. C. (2004). A record of seahorse species (Family Syngnathidae) in East Malaysia, with notes on their conservation. *Malayan Nature Journal* **56**, 409–420.

Choo, C. K. & Liew, H. C. (2005). Exploitation and trade in seahorses in Peninsular Malaysia. *Malayan Nature Journal* **57**, 57–66.

Chopin, F. S. & Arimoto, T. (1995). The condition of fish escaping from fishing gears - a review. *Fisheries Research* **21**, 315–327.

Chuenpagdee, R., Liguori, L., Palomares, M. L. D., Pauly, D., (2006). Bottom-up, Global Estimates of Small-Scale Marine Fisheries Catches. In *Fisheries Centre Research Reports*, Vol.14 (8), pp. 1-94. Vancouver, British Columbia: Fisheries Centre, The University of British Columbia.

Costello, C., Gaines, S. D. & Lynham, J. (2008). Can catch shares prevent fisheries collapse? *Nature* **321**, 1678–1681.

Curtis, J. M. R., & Vincent, A. C. J. (2006). Life history of an unusual marine fish: survival, growth and movement patterns of *Hippocampus guttulatus* Cuvier 1829. *Journal of Fish Biology* **68**, 707–733. doi:10.1111/j.1095-8649.2006.00952.x

Davies, R., Cripps, S. J., Nickson, A. & Porter, G. (2009). Defining and estimating global marine fisheries bycatch. *Marine Policy* **33**, 661-672.

Dulvy, N. K., & Reynolds, J. D. (2002). Predicting extinction vulnerability in skates.

Conservation Biology **16**, 440–450.

Dulvy, N. K., Sadovy, Y. & Reynolds J. D. (2003). Extinction vulnerability in marine populations. *Fish and Fisheries* **4**, 25–64.

Dulvy, N. K., Ellis, J. R., Goodwin, N. B., Grant, A., Reynolds, J. D. & Jennings, S. (2004). Methods of assessing the extinction risk in marine fishes. *Fish and Fisheries* **5**, 255–276.

Duplisea, D. E., Jennings, S., Warr, K. J. & Dinmore, T. A. (2002). A size-based model of the impacts of bottom trawling on benthic community structure. *Canadian Journal of Fisheries and Aquatic Sciences* **59**, 1785–1795.

Estalles, M., Coller, N. M. & Perier, M. R. (2011). Skates in the demersal trawl fishery of San Matías Gulf, Patagonia: species composition, relative abundance and maturity stages. *Aquatic Living Resources* **24**, 193-199. doi:10.1051/alr/2011119

FAO (2013). Report of the fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species. *Fisheries and Aquaculture Report No R1032*. Rome: Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations.

FAO (2014). *The State of World Fisheries and Aquaculture 2014*. Rome: Fisheries and Aquaculture Department, Food and Agriculture Organisation of the United Nations.

Foster, S. J. & Vincent, A. C. J. (2004). Life history and ecology of seahorses: implications for conservation and management. *Journal of Fish Biology* **65**, 1–61. doi:10.1111/j.1095-8649.2004.00429.x

Foster, S. J. & Vincent, A. C. J. (2005). Enhancing Sustainability of the International Trade in Seahorses with a Single Minimum Size Limit. *Conservation Biology* **19**, 1044–1050. doi: 10.1111/j.1523-1739.2005.00192.x

Foster, S. J. & Vincent, A. C. J. (2010*a*). Tropical shrimp trawl fisheries: Fishers' knowledge of and attitudes about a doomed fishery. *Marine Policy* **34**, 437–446.

Foster, S. J. & Vincent, A. C. J. (2010*b*). Using life-history information to assess potential effects of shrimp trawling on small fishes. *Journal of Fish Biology* **76**, 2434–2454.

Foster, S. J. & Vincent, A. C. J. (2012). Advice in spite of great uncertainty: assessing and addressing bycatch of small fishes with limited data using *Stellifer illecebrosus* as a case study. *Aquatic Conservation: Marine and Freshwater Ecosystems* **22**, 639–651.

Freese L., Auster, P. J., Heifetz, J. & Wing, B.L. (1999). Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska. *Marine Ecology Progress Series* **189**, 119–26.

Froese, R. (2004). Keep it simple: three indicators to deal with overfishing. *Fish and Fisheries* **5**, 86–91.

Funge-Smith, S., Lindebo, E. & Staples, D. (2005). Asian fisheries today: The production and use of low value/trash fish from marine fisheries in the Asia-Pacific region. *RAP Publication 2005/16*. Bangkok: The Asia-Pacific Fishery Commission.

Funge-Smith, S., Briggs, M. & Miao, W. (2012). Regional overview of fisheries and aquaculture in Asia and the Pacific 2012. *RAP Publication 2012/26*. Bangkok: The Asia-Pacific Fishery Commission.

Giles, B. G., Ky, T. S., Hoang, D. H. & Vincent, A. C. J. (2006). The catch and trade of seahorses in Vietnam. *Biodiversity and Conservation* **15**, 2497–2513.

Goñi, R. (1998). Ecosystem effects of marine fisheries: an overview. *Ocean and Coastal Management* **40**, 37–64.

Hall, M. A. (1997). On bycatches. *Reviews in Fish Biology and Fisheries* **6**, 319–352.

Hall, M. A., Alverson, D. L. & Metuzals, K. I. (2000). By-catch: problems and solutions. *Marine Pollution Bulletin* **41**, 204–219.

Hall, S. J. & Mainprize, B. M. (2005). Managing by-catch and discards: how much progress are we making and how can we do better? *Fish and Fisheries* **6**, 134–155.

Harasti, D., Martin-Smith, K. & Gladstone, W. (2012). Population dynamics and life history of a geographically restricted seahorse, *Hippocampus whitei*. *Journal of Fish Biology* **81**, 1297–1314. doi:10.1111/j.1095-8649.2012.03406.x

Hiew, K., Saad, J. & Gopinath, N. (2013). *Coral Triangle Initiative: Ecosystem Approach to Fisheries Management (EAFM): Country Position Paper—Malaysia.* Honolulu, HI: The USAID Coral Triangle Support Partnership.

Hill, B. J., & Wassenberg, T. J. (2000). The probable fate of discards from prawn trawlers fishing near coral reefs: A study in the northern Great Barrier Reef, Australia. *Fisheries Research* **48**, 277–286.

Hutchings, J.A. (2003). Life histories of fish. In *Handbook of Fish Biology and Fisheries*, Vol. 1 (Hart, P. J. B. & Reynolds, J. D., eds), pp. 149–174. Oxford: Blackwell Science.

Hutchings, J. A. & Reynolds, J. D. (2004). Marine Fish Population Collapses: Consequences for Recovery and Extinction Risk. *BioScience* **54**, 297–309. doi: http://dx.doi.org/10.1641/0006-3568(2004)054[0297:MFPCCF]2.0.CO;2

Hutchings, J. A. & Baum, J. K. (2005). Measuring marine fish biodiversity: temporal changes in abundance, life history and demography. *Philosophical Transactions of the Royal Society B: Biological Sciences* **360**, 315-338. doi:10.1098/rstb.2004.1586

Hutchings, J. A. & Rowe, S. (2008). Consequences of sexual selection for fisheries-induced evolution: an exploratory analysis. *Evolutionary applications* **1**, 129-136.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J. & Warner, R. R. (2001) Historical overfishing and the recent collapse of coastal ecosystems. *Science* **293**, 629–638.

Jennings, S., & Kaiser, M. J. (1998). The effects of fishing on marine ecosystems. *Advances in Marine Biology* **34**, 201–352.

Jennings, S., Reynolds, H. D. & Mills, S. C. (1998). Life history correlates of responses to fisheries exploitation. *Proceedings of the Royal Society of London B* **265**, 333–339.

Jennings, S., Reynolds, J. D. & Polunin, N. V. C. (1999). Predicting the Vulnerability of Tropical Reef Fishes to Exploitation with Phylogenies and Life Histories. *Conservation Biology* **13**, 1466–1475.

Johannes, R. E. 1998. The case for data-less marine resource management: examples from tropical nearshore finfisheries. *Trends in Ecology & Evolution* **13**, 243–246.

Kelleher, K. (2005). Discards in the World's Marine Fisheries. *Fisheries Technical Paper No 470*. Rome: Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations.

King, M. (2007). *Fisheries Biology, Assessment and Management,* 2nd edn. Oxford: John Wiley & Sons.

Kitchell, J. F., Kaplan, I. C., Cox, S. P., Martell, S. J. D., Essington, T. E., Boggs, C. H., and Walters, C. J. (2004). Ecological and economic components of alternative fishing methods to reduce by-catch of marlin in a tropical pelagic ecosystem. *Bulletin of Marine Science* **74**, 607–619.

Kumar, A. B. & Deepthi, G. R. (2006). Trawling and by-catch: implications on marine ecosystems. *Current Science* **90**, 922–931.

Kuparinen, A., Kuikka, S. & Merilä, J. (2009). Estimating fisheries-induced selection: traditional gear selectivity research meets fisheries-induced evolution. *Evolutionary Applications* **2**, 234–243. doi:10.1111/j.1752-4571.2009.00070.x

Laksanawimol, P., Petpiroon, S. & Damrongphol, P. (2013). Trade of Seahorses, *Hippocampus* spp. (Actinopterygii: Syngnathiformes: Syngnathidae), on the East Coast of the Gulf of Thailand. *Acta Ichthyologica Et Piscatoria* **43**, 229–235.

La Manach, F., Gough, C., Harris, A., Humber, F., Harper, S. & Zeller, D. (2012). Unreported fishing, hungry people and political turmoil: the recipe for a food security crisis in Madagascar? *Marine Policy* **36**, 218–225.

Lewison, R. L., Crowder, L. B., Read, A. J. & Freeman, S. A. (2004). Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology & Evolution* **19**, 598–604.

Lim, A., Chong, V. C., Wong, C. S. & Choo, C. K. (2011). Diversity, habitats and conservation threats of syngnathid (Syngnathidae) fishes in Malaysia. *Tropical Zoology* **24**, 193–222.

Lobo, A. S., Balmford, A., Arthur, R. & Manica, A. (2010). The bycatch problem: effects of commercial fisheries on non-target species in India. *Resonance* **12**, 60–70.

Lourie, S. A. (2003). *Measuring Seahorses.* Project Seahorse Technical Report No **4**. Project Seahorse, Fisheries Centre, The University of British Columbia. 15 pp.

Lourie, S. A., Pritchard, J. C., Casey, S. P., Truong, S. K., Hall, H. J. & Vincent, A. C. J. (1999). The taxonomy of Vietnam's exploited seahorses (family Syngnathidae). *Biological Journal of the Linnean Society* **66**, 231–256.

Lourie, S. A., Foster, S. J., Cooper, E. W. T., & Vincent, A. C. J. (2004). *A Guide to the*

Identification of Seahorses. Project Seahorse and TRAFFIC North America. Washington D.C.: University of British Columbia and World Wildlife Fund.

Martin-Smith, K. M. & Vincent, A. C. J. (2006). Exploitation and trade of Australian seahorses, pipehorses, sea dragons and pipefishes (Family Syngnathidae). *Oryx* **40**, 141–151.

McPherson, J. M. & Vincent, A. C. J. (2004). Assessing East African trade in seahorse species as a basis for conservation under international controls. *Aquatic Conservation: Marine and Freshwater Ecosystems* **14**, 521–538.

Meeuwig, J. J., Hoang, D. H., Si Ky, T., Job, S. D., Vincent, A. C. J. (2006). Quantifying non-target seahorse fisheries in central Vietnam. *Fisheries Research* **81**, 149–157.

Morgan, S. K. & Vincent, A. C. J. (2013). Life-history reference points for management of an exploited tropical seahorse. *Marine and Freshwater Research* **64**, 185–200. doi:http://dx.doi.org/10.1071/MF12171

Mundy-Taylor V. & Crook, V. (2013). *Into the deep: Implementing CITES measures for commercially-valuable sharks and manta rays.* Cambridge: TRAFFIC International.

Murugan, A., Dhanya, S., Rajagopal, S. & Balasubramanian, T. (2008). Seahorses and pipefishes of the Tamil Nadu coast. *Current Science* **95**, 253–260.

Murugan, A., Dhanya, S., Sreepada, R. A., Rajagopal, S. & Balasubramanian, T. (2009). Breeding and mass-scale rearing of three spotted seahorse, *Hippocampus trimaculatus* Leach under captive conditions. *Aquaculture* **290**, 87–96. doi:10.1016/j.aquaculture.2009.01.033

Murugan, A., Dhanya, S., Sarcar, A. B., Naganathan, V., Rajagopal, S. & Balasubramanian, T. (2011). Fishery biology, demography of the three spotted seahorse*, Hippocampus trimaculatus* inhabiting Gulf of Mannar region, Southeast coast of India. *Indian Journal of Geo Marine Sciences* **40**, 411–423.

Nadeau, J. L., Curtis, J. M. R. & Lourie, S. A. (2009). Preservation causes shrinkage in seahorses: implications for biological studies and for managing sustainable trade with minimum size limits. *Aquatic Conservation: Marine and Freshwater Ecosystems* **19**, 428–438. doi:10.1002/aqc.1002

Nédélec, C. & Prado, J. (1990). Definition and classification of fishing gear categories. *Fisheries Technical Paper No 222*. Rome: Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations.

Nguyen, V. L. & Do, H. H. (1996). Biological parameters of two exploited seahorse species in a Vietnamese fishery. In *Proceedings of the 1st International Conference in Marine Conservation*. Hong Kong.

O'Donnell, K. P., Molloy, P. P. & Vincent, A. C. J. (2010) Comparing Fisher Interviews, Logbooks, and Catch Landings Estimates of Extraction Rates in a Small-Scale Fishery. *Coastal Management* **40**, 594–611.

Pauly, D. (1996). Fleet-operational, economic, and cultural determinants of bycatch uses in Southeast Asia. In *Solving Bycatch: Considerations for Today and Tomorrow*, Report No. 96-03. Fairbanks, AK: Sea Grant College Program, University of Alaska.

Pauly, D., Christensen, V., Guenette, S., Pitcher, T. J., Sumaila, U. R., Walters, C. J. & Zeller, D. (2002). Towards sustainability in world fisheries. *Nature* **418**, 689-695.

Pauly, D., Watson, R. & Alder, J. (2005). Global trends in world fisheries: impacts on marine ecosystems and food security. *Philosophical Transactions of the Royal Society B: Biological Sciences* **360**, 5–12.

Perante, N. C., Pajaro, M. G., Meeuwig J. J. & Vincent A. C. J. (2002). Biology of a seahorse species *Hippocampus comes* in the central Philippines. *Journal of Fish Biology* **60**, 821–837.

doi:10.1006/jfbi.2002.1878

Perry, A. L., Lunn, K. E. & Vincent, A. C. J. (2010). Fisheries, large-scale trade, and conservation of seahorses in Malaysia and Thailand. *Aquatic Conservation: Marine and Freshwater Ecosystems* **20**, 464–475.

Reynolds, J. D., Dulvy, N. K., Goodwin, N. B. & Hutchings, J. A. (2005). Biology of extinction risk in marine fishes. *Proceedings of the Royal Society of London B* **272**, 2337–2344.

Roberts, C. M. & Hawkins, J. P. (1999). Extinction risk in the sea. *Trends in Ecology and Evolution* **14**, 241–246.

Rowe, S. & Hutchings, J. A. (2003). Mating systems and the conservation of commercially exploited marine fish. *TRENDS in Ecology and Evolution* **18**, 567–572. doi:10.1016/j.tree.2003.09.004

Rueda, M. & Defeo, O. (2003). Linking fishery management and conservation in a tropical estuarine lagoon: biological and physical effects of an artisanal fishing gear. *Estuarine, Coastal and Shelf Science* **56**, 935–942. doi:10.1016/S0272-7714(02)00298-6

Sadovy, Y. (2001). The threat of fishing to highly fecund fishes. *Journal of Fish Biology* **59**, 90–108. doi:10.1006/jfbi.2001.1760

Salin, K. R., Yohannan, T. M. & Mohanakumaran, N. C. (2005). Fisheries and trade of seahorses, *Hippocampus* spp., in southern India. *Fisheries Management and Ecology* **12**, 269–273.

Scales, H. (2009). *Poseidon' s Steed: The Story of Seahorses, From Myth to Reality.* New York, NY: Penguin Group Inc.

Soykan, C. U., Moore, J. E., Zydelis, R., Crowder, L. B., Safina, C. & Lewison, R. L. (2008). Why study bycatch? An introduction to the theme selection on fisheries bycatch.

Endangered Species Research **5**, 91–102.

Sparre, P., Ursin, E. & Venema, S. C. (1989). *Introduction to Tropical Fish Stock Assessment*. Food & Agriculture Organization Fisheries Technical Paper No **306**.

Stewart, K., Johnson, C. & Godfrey, M. H. (2007). The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. *Herpetological Journal* **17**, 123–128.

Stobutzki, I. C., Miller, M. J., Jones, P. & Salini, J. P. (2001*a*). Bycatch diversity and variation in a tropical Australian penaeid fishery; the implications for monitoring. *Fisheries Research* **53**, 283–301.

Stobutzki, I., Miller, M. & Brewer, D. (2001*b*). Sustainability of fishery bycatch: a process for assessing highly diverse and numerous bycatch. *Environmental Conservation* **28**:167–181.

Sumaila, U. R., Khan, A. S., Dyck, A. J., Watson, R., Munro, G., Tydemers, P. & Pauly, D. (2010). A bottom-up re-estimation of global fisheries subsidies. *Journal of Bioeconomics* **12**, 201-225. doi: 10.1007/s10818-010-9091-8

Swartz, W., Sala, E., Tracey, S., Watson, R. & Pauly, D. (2010). The spatial expansion and ecological footprint of fisheries (1950 to present). *Public Library of Science ONE* **5**, e15143.

Sylwester, J. G. (2014). Fishers of men: the neglected effects of environmental depletion on labor trafficking in the Thai fishing industry. *Pacific Rim Law and Policy Journal* **23**, 423–461.

Tacon, A. G. & Metian, M. (2009). Fishing for feed or fishing for food: increasing global competition for small pelagic forage fish. *AMBIO: A Journal of the Human Environment* **38**, 294–302.

Teixeira, R. L., & Musick, J. A. (2001). Reproduction and food habits of the lined seahorse, *Hippocampus erectus* (Teleostei: Syngnathidae) of Chesapeake Bay, Virginia. *Revista Brasileira de Biologia* **61**, 79–90.

Vincent, A. C. J. (1990). *Reproductive ecology of seahorses*. PhD thesis, Cambridge, University of Cambridge. United Kingdom.

Vincent, A.C.J. (1996). *The International Trade in Seahorses*. Cambridge: TRAFFIC International.

Vincent, A. C. J., Evans, K. L. & Marsden, A. D. (2005). Home range behaviour of the monogamous Australian seahorse, *Hippocampus whitei*. *Environmental Biology of Fishes* **72**, 1–12.

Vincent, A. C. J., Giles, B. G., Czembor, C. A. & Foster, S. J. (2011*a*). Trade in seahorses and other syngnathids in countries outside Asia (1998-2001). In *Fisheries Centre Research Reports*, Vol. 19(1), pp. 1–181. Vancouver, British Columbia: Fisheries Centre, The University of British Columbia.

Vincent, A. C. J., Foster, S. J. & Koldewey, H. J. (2011*b*). Conservation and management of seahorses and other Syngnathidae. *Journal of Fish Biology* **78**, 1681–1724.

Vincent, A. C. J., Sadovy de Mitcheson, Y. J., Fowler, S. L. & Lieberman, S. (2013). The role of CITES in the conservation of marine fishes subject to international trade. *Fish and Fisheries*. doi:10.1111/faf.12035

Walters, C. J. & Martell, S. J. D. (2004). *Fisheries Ecology and Management*. Princeton NJ: Princeton University Press.

Walting, L. (2014). Trawling exerts big impacts on small beasts. *Proceedings of the National Academy of Sciences USA* **111**, 8704–8705.

Watson, R. A., Cheung, W. W. L., Anticamara, J. A., Sumaila, R. U., Zeller, D. & Pauly, D.

(2013). Global marine yield halved as fishing intensity redoubles. *Fish and Fisheries* **14**, 493–503.

Wilson, S. M., Raby, G. D., Burnett, N. J., Hinch, S. G. & Cooke, S. J. (2014). Looking beyond the mortality of bycatch: sublethal effects of incidental capture on marine animals. *Biological Conservation* **171**, 61–72.

Winemiller, K. O. (2005). Life history strategies, population regulation, and implications for fisheries management. *Canadian Journal of Fisheries and Aquatic Sciences* **62**, 872–885.

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J. and Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science* **314**, 787–790.

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., Fogarty, M. J., Fulton, E. A., Hutchings, J. A., Jennings, S., Jensen, O. P., Lotze, H. K., Mace, P. M., McClanahan, T. R., Minto, C., Palumbi, S. R., Parma, A. M., Ricard, D., Rosenberg, A. A., Watson, R. & Zeller, D. (2009). Rebuilding global fisheries. *Science*, **325**, 578–585.

Worm, B. & Branch, T.A. (2012). The future of fish. *Trends in Ecology and Evolution*, **27**, 594–599.

Electronic References

CITES (2004). Notification to the Parties No. 2004/033. Concerning Trade in Seahorses. Available at http://www.cites.org/eng/notif/2004/033.pdf/ (last accessed 29 June 2014).

CITES (2013). Notification to the Parties No. 2013/012. Amendment to Appendices I and II of the Convention. Available at http://www.cites.org/sites/default/files/eng/notif/2013/E-Notif-

2013-012.pdf/ (last accessed 29 June 2014).

Froese, R. & Pauly, D. (Eds) (2014). *FishBase.* Version 06/2014. Available at http://www.fishbase.org, (last accessed 29 June 2014).

IUCN (2013). *IUCN Red List of Threatened Species.* Version 2013.2. Available at http://www.iucnredlist.org/ (last accessed 31 March 2014).

IUCN (2014). *IUCN Red List of Threatened Species*. Version 2014.1. Available at http://www.iucnredlist.org/ (last accessed 29 June 2014).

R Core Development Team. (2013). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. Available at http://www.R-project.org/ (last accessed 31 March 2014)

UNEP-WCMC (2012). *CITES Trade Database.* Available at http://www.unep-wcmc.org/citestrade/trade.cfm/ (last accessed 31 March 2014)

Appendices

Appendix A List of all countries and sources examined for bycatch information

The 36 countries that were examined in this report, and the associated published and unpublished sources where data on catch-per-unit-effort (CPUE), estimated total annual volumes in bycatch, CITES international trade records, and changes in catch charactaristics (specifically change in catch volume and change in catch mean size) over time were extracted from. The * indicates instances where CPUEs or total annual volumes were reported but were not specific to any gear, so data was not used in gear type analysis. Also included is the method of data collection – (1) *in situ* interviews with trade participants or those with local knowledge of the seahorse trade ("interviews"), (2) official data collected by government agencies detailing either the catch or trade of seahorses ("official data"), (3) personal communications sent to Project Seahorse scientists ("pers. comm."), and (4) *in situ* direct monitoring of catches or landings of bycaught seahorses ("surveys").

Country	**Study**	**Data Collection Method**	**CPUE**	**Total Annual Volume**	**CITES Data**	**Change in catch characteristics over time**	
						Catches	***Mean Size***
Argentina	Vincent *et al.*, 2011a	Pers. comm.					
Australia	Vincent *et al.*, 2011a	Official data	Reported	Reported*			
Bangladesh	B. Giles, Project Seahorse,	Interviews	Reported	Reported		Reported	

Country	Study	Data Collection Method	CPUE	Total Annual Volume	CITES Data	Change in catch characteristics over time	
						Catches	***Mean Size***
	unpublished data						
Belize	Baum & Vincent, 2005	Interviews				Reported	
	Vincent *et al*., 2011a	Interviews	Reported	Reported			
	Foster *et al.*, in review	Official data			Reported		
Brazil	Vincent *et al*., 2011a	Interviews					
China	Project Seahorse, unpublished data	Interviews	Reported*				
Costa Rica	Baum & Vincent, 2005	Interviews				Reported	
	Vincent *et al*., 2011a	Interviews	Reported				
Croatia	Vincent *et al*., 2011a	Pers. comm.					
Ecuador	Baum & Vincent, 2005	Interviews				Reported	
	Vincent *et al*., 2011a	Interviews	Reported	Reported			
	Foster *et al*., in review	Official data			Reported		
France	Vincent *et al*., 2011a	Pers. comm.					

Country	Study	Data Collection Method	CPUE	Total Annual Volume	CITES Data	Change in catch characteristics over time	
						Catches	***Mean Size***
Guatemala	Baum & Vincent, 2005	Interviews				Reported	
	Vincent *et al*., 2011a	Interviews	Reported	Reported			
Honduras	Baum & Vincent, 2005	Interviews				Reported	
	Vincent *et al*., 2011a	Interviews	Reported	Reported			
India	A. Perry, Project Seahorse, unpublished data	Interviews	Reported	Reported		Reported	Reported
	Salin *et al.*, 2005	Surveys	Reported	Reported			
	Murugan *et al.*, 2011	Surveys	Reported	Reported*			
Indonesia	A. Perry, Project Seahorse, unpublished data	Interviews	Reported			Reported	
Japan	Project Seahorse, unpublished data	Pers. comm.					
Kenya	Vincent *et al*., 2011a	Interviews					
Malaysia	Choo & Liew, 2005	Interviews	Reported	Reported			

Country	Study	Data Collection Method	CPUE	Total Annual Volume	CITES Data	Change in catch characteristics over time	
						Catches	***Mean Size***
	Foster, *et al.*, in review	Official data			Reported		
	J. Lawson, Project Seahorse, pers. obs.	Interviews	Reported				
	Perry *et al.*, 2010		Reported	Reported		Reported	Reported
Mexico	Baum & Vincent, 2005	Interviews	Reported	Reported		Reported	
	Foster *et al.*, in review	Official data			Reported		
Mozambique	Vincent *et al.*, 2011a	Pers. comm.					
New Zealand	Foster *et al.*, in review	Official data			Reported*		
	Vincent *et al.*, 2011a	Official data	Reported*				
Nicaragua	Baum & Vincent, 2005	Interviews			Reported	Reported	
	Vincent *et al.*, 2011a	Interviews	Reported	Reported			
Nigeria	Vincent *et al.*, 2011a	Pers. comm.					
Pakistan	A. Perry, Project Seahorse, unpublished data	Interviews	Reported	Reported	Reported*	Reported	

Country	Study	Data Collection Method	CPUE	Total Annual Volume	CITES Data	Change in catch characteristics over time	
						Catches	***Mean Size***
Panama	Baum & Vincent, 2005	Interviews				Reported	
	Foster *et al.*, in review	Official data			Reported		
	Vincent *et al.*, 2011a		Reported	Reported			
Peru	Alfaro-Shigueto & Mangel, 2011	Surveys	Reported	Reported			
	Baum & Vincent, 2005	Interviews				Reported	
	Foster *et al.*, in review	Official data			Reported		
	Vincent *et al.*, 2011a	Interviews	Reported				
Philippines	Foster *et al.*, in review	Official data			Reported		
	M. Pajaro, Project Seahorse, unpublished data	Interviews	Reported	Reported			
Portugal	Vincent *et al.*, 2011a	Pers. comm.					
Senegal	A. Cisneros-Montemayor, Project Seahorse, unpublished	Interviews	Reported	Reported		Reported	Reported

Country	Study	Data Collection Method	CPUE	Total Annual Volume	CITES Data	Change in catch characteristics over time	
						Catches	***Mean Size***
	data						
	Foster *et al.*, in review	Official data			Reported		
South Korea	Choi *et al.*, 2012	Interviews					
Spain	Vincent *et al*., 2011a	Pers. comm.					
Taiwan	Project Seahorse, unpublished data	Interviews					
Tanzania	McPherson & Vincent, 2004	Interviews				Reported	
	Vincent *et al*., 2011a	Interviews	Reported	Reported			
Thailand	Foster *et al.*, in review	Official data			Reported		
	Laksanawimol *et al*., 2013	Interviews	Reported				
	Perry *et al.*, 2010	Interviews	Reported	Reported		Reported	Reported
	T-C. Kuo, Project Seahorse, unpublished data	Interviews	Reported	Reported			
Turkey	Vincent *et al*., 2011a	Pers. comm.					

Country	Study	Data Collection Method	CPUE	Total Annual Volume	CITES Data	Change in catch characteristics over time	
						Catches	***Mean Size***
USA	Baum *et al.*, 2003	Interviews	Reported				
	Vincent *et al*., 2011a	Official data	Reported	Reported			
Vietnam	Foster *et al.*, in review	Official data			Reported		
	Giles *et al.*, 2006	Interviews	Reported	Reported		Reported	
	Meeuwig *et al.*, 2006	Surveys	Reported				

Appendix B List of catch-per-unit-effort (CPUE) and total estimated volume of seahorses in bycatch reported by country

B.1 Gear type-specific median catch-per-unit-effort and associated countries, sources and sample sizes

The 24 countries and associated sources for which catch-per-unit effort (CPUE; seahorses^{-1} *vessel^{-1}*day^{-1}) was estimated based on *in situ* interviews, official data, or *in situ* direct monitoring of catches/landings. CPUE was calculated for five categories of fishing gear, and are reported here to two decimal places. Where ranges or multiple estimates of CPUE were provided, I took the median (M) of these values, and reported both the number of values used to calculate the median (N1) as well as the total number of fishers or vessels that generated the values (N2), where information was provided (otherwise indicated with N/S). Also included are median CPUEs for other gears (trap, falling gear, dredge and hook/line) and unknown gear types where few estimates were available.

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	N1	N2	M	N1	N2	M	N1	N2	M	N1	N2	M	N1	N2	M	N1	N2
Australia	Vincent *et al.*, 2011a	0.01	2	N/S															
Bangladesh	B. Giles, Project	0.01	5	5	0.65	8	1												

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**
	Seahorse, unpublished data						0												
Belize	Vincent *et al*., 2011a	11.50	2	3										0.92	10	13			
China	Project Seahorse, unpublished data																0.33	2	N/S
Costa Rica	Vincent *et al*., 2011a	4.76	1	3															
Ecuador	Vincent *et al*., 2011a	0.69	2	N/S															
Guatemala	Vincent *et al*., 2011a	0.37	4	9	0.02	2	3												
Honduras	Vincent *et al*., 2011a	7.23	4	1	0.34	2	8												

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	N1	N2	M	N1	N2	M	N1	N2	M	N1	N2	M	N1	N2	M	N1	N2
				9															
India	A. Perry, Project Seahorse, unpublished data	0.90	26	N/S	1.5	2	10	4.74	13	N/S	1.5	2	10						
	Salin *et al.*, 2005	23.21	24	120															
	Murugan *et al.*, 2011	3.20	3	N/S															
Indonesia	A. Perry, Project Seahorse, unpublished data				0.52	2	N/S	5.71	2	1	4.15	5	3						

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**
Kenya	Vincent *et al*., 2011a																0.10	N/S	N/S
Malaysia	Choo & Liew, 2005	0.48	12	N/S															
	J. Lawson, Project Seahorse, pers. obs.				0.33	8	5				0.02	1	1						
	Perry *et al*., 2010	0.47	13	55	0.85	14	9										3.57	2	2
Mexico	Vincent *et al*., 2011a	0.33	5	63													0.05	2	1
New	Vincent *et al*., 2011a																0.08	2	9

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**
Zealand																		2	
Nicaragua	Vincent *et al*., 2011a	6.17	3	16															
Pakistan	A. Perry, Project Seahorse, unpublished data	0.33	3	12	1.29	1	4												
Panama	Vincent *et al*., 2011a	0.24	4	N/S															
Peru	Alfaro-Shigueto & Mangel, 2011				1.72	1	12				14.29	5	3						
	Vincent *et al*., 2011a	0.86	5	4	3.0	2	2												
Philippines	M. Pajaro, Project	3	1	1				2	1	5				2	1	2			

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**
	Seahorse, unpublished data			1															
Senegal	A. Cisneros-Montemayor, Project Seahorse, unpublished data	23.22	4	4	0.04	28	28	0.21	7	7	0.73	4	4				0.29	20	20
Tanzania	Vincent *et al.*, 2011a	0.42	2	1							1	7	5				0.07	3	3
Thailand	Laksanawimol *et al.*, 2013	3.87	6	50													0.56	11	65
	Perry *et al.*, 2010	0.77	24	14							1.50	2	1	31.48	2	2			
	T-C. Kuo, Project	0.76	8	5	0.49	7	2												

Country	Study	Trawl net			Gill/Entangling net			Seine net			Surround net			Scoop net			Other gears/unknown		
		M	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**	**M**	**N1**	**N2**
	Seahorse, unpublished data			5			1												
USA	Baum *et al.*, 2003	9.64	1	445															
	Vincent *et al.*, 2011a	0.02	4	N/S															
Vietnam	Giles *et al.*, 2006	1.55	12	308															
	Meeuwig *et al.*, 2006	1.24	4	16															

B.2 Gear type-specific total estimated volumes of bycaught seahorses, with associated countries and fleet sizes

Estimated annual total catch (in number of individual seahorses) for four gear types (and for volumes where gear type was not specified), across the 18 countries where CPUE data was extrapolated by authors. Also included with the total volume is the number of vessels (as reported by authors) for which CPUEs were scaled-up in order to estimate total volumes. Countries are listed alphabetically.

Country	Study	Trawl net		Gill/Entangling net		Seine net		Scoop net		Unknown gear	
		Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels
Bangladesh	B. Giles, Project Seahorse, unpublished data			201,600	N/S			1,020,337	2422		
Belize	Vincent *et al*., 2011a	18402	3								
Ecuador	Vincent *et al*., 2011a	51,000	200								
Guatemala	Vincent *et al*., 2011a	37,960	107								

Country	Study	Trawl net		Gill/Entangling net		Seine net		Scoop net		Unknown gear	
		Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels
Honduras	Vincent *et al.*, 2011a	267,288	130								
India	Salin *et al.*, 2005	3,209,597	N/S								
	Murugan *et al.*, 2011									104,018	N/S
Malaysia	Choo & Liew, 2005	565,640	437								
	Perry *et al.*, 2010	919,747	6035								
Mexico	Baum & Vincent, 2005	330,864	N/S								
New Zealand	Vincent *et al.*, 2011a									1,115	N/S
Nicaragua	Vincent *et al.*, 2011a	187,547	87								
Pakistan	A. Perry, Project	83,400	300								

Country	Study	Trawl net		Gill/Entangling net		Seine net		Scoop net		Unknown gear	
		Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels
	Seahorse, unpublished data										
Panama	Vincent *et al.*, 2011a	20,260	232								
Peru	Alfaro-Shigueto & Mangel, 2011			25,000	3						
Philippines	M. Pajaro, Project Seahorse, unpublished data	553,350	1,085			159,975	395	79,950	205		
Senegal	A. Cisneros-Montemayor, Project Seahorse, unpublished data	114,023	91							371,567	7257
Tanzania	Vincent *et al.*,									42,000	

Country	Study	Trawl net		Gill/Entangling net		Seine net		Scoop net		Unknown gear	
		Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels	Total	No. of Vessels
	2011a										
Thailand	Perry *et al.*, 2010	1,632,626	5,163					596,040	44		
	T-C. Kuo, Project Seahorse, unpublished data	708,650	3,539	32,691	1670						
Vietnam	Giles *et al.*, 2006	2,067,625	9,120								
	Meeuwig *et al.*, 2006	49,250	160								