IS BYCATCH A BIG PROBLEM FOR SMALL FISH? ASSESSING AND ADDRESSING THE IMPACTS OF TROPICAL SHRIMP TRAWLING ON SMALL FISH SPECIES

by

SARAH JANE FOSTER
B.Sc. (Hons), Dalhousie University, 1998
M.Sc., Dalhousie University, 2001

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF
THE REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

in

THE FACULTY OF GRADUATE STUDIES
(Resource Management and Environmental Studies)

THE UNIVERSITY OF BRITISH COLUMBIA
(Vancouver)

December 2009

Abstract

My research, focused on the industrial shrimp fishery in southern Gulf of California, aimed to determine whether tropical shrimp trawling is a problem for viability of incidentally captured small fish species.

My first objective was to use life history information to evaluate possible impacts on four small fish species from their incidental capture. I applied length based indicators and qualitative criteria to information on captured sizes, reproduction, and distributions across the study site. My results suggested potential for overfishing silver stardrum and bigscale goatfish, largely because most sampled individuals were immature. Silver stardrum may be particularly affected because its occurrence and density declined during the trawling season whereas goatfish apparently recruited to the study area. Most sampled sandperch were mature, suggesting greater resilience to trawling. In contrast, sampled lumptail searobin, although mature, had not yet spawned, indicating potential for adverse fishing effects.

Because human behaviour affects the success of fisheries management, my second objective was to shed light on the social dimensions of tropical shrimp fisheries management. My interviews with industrial trawl fishers suggested that proper enforcement and reliable governance are essential for a sustainable fishery. If enforcement were strong, then most fishers would support trawl free areas. The effort data I gathered point to areas where protection might be socially acceptable.

My third objective considered the biological appropriateness of trawl closures for small fishes. The divergent distributions of bigscale goatfish and silver stardrum, just two of many small species in bycatch, implied that trawl restrictions would have to cover many depths and latitudes. Further, although my matrix model was of limited use for assessing population status of silver stardrum, it clearly indicated that precautionary management should focus on increasing survival of younger fish. This could be achieved with trawl closures where smaller fish live.

While the approaches I used identified small fishes that might be vulnerable to trawling, they are too data intensive to be viable for the hundreds of such species in bycatch, and too inconclusive

to confirm impact. It may be necessary to apply precautionary methods such as trawl closures to avoid potential effects of indiscriminate trawling.

Table of contents

List of tables

List of figures

Acknowledgements

I would like to start by thanking my research supervisor, Dr. Amanda Vincent. From the hatching of the idea for this research over a margarita on the beach, to working long hours finalising the draft when you were on leave, you have been a dedicated mentor, colleague and friend for the duration of this amazing journey (indeed, well before it began). You have granted me some amazing opportunities, and this research is no exception. Thank you for believing in me, but mostly, for giving me opportunities to believe in myself. I look forward to many more years of collaboration as we continue to advance marine conservation, and our love of margaritas!

My sincere thanks also to each of my committee members: Dr. Scott Hinch, Dr. Steve Martell and Dr. Daniel Pauly. From inspiring discussions in committee meetings, to providing individual advice and guidance on my chapters, you have each played an important role in my training as a scientist, and making my research ideas a reality.

This research was made possible with generous financial support from The International Development Research Centre (Canada), EJLB Foundation, IODE, Natural Sciences and Engineering Research Council of Canada and The University of British Columbia grants and fellowships. I also thank The John G. Shedd Aquarium in Chicago and Guylian Chocolates Belgium for their assistance through our partnerships for marine conservation. Additional logistical support came from the RMES graduate program coordinators Lisa Belanger and Barry Wayne, the UBC Fisheries Centre (especially Janice Doyle), and the University of British Columbia. CICIMAR was supported by grants from CONACyT-SAGARPA (2003-02-157, SIP 20090922), EDI and COFAA.

This study could not have happened without my collaboration with Dr. Francisco Arreguín-Sanchez and his team at the Centro Interdisciplinario de Ciencias Marinas (CICIMAR) in La Paz, Baja California Sur, Mexico, and Escuela Nacional de Ingeniería Pesquera (ENIP) in San Blas, Nayarit, Mexico. Thank you for supporting my research ideas, and giving me the opportunity to realise them in Mexico. I hope to find ways we can work together in the future. In particular, I would like to extend my sincere thanks to my colleagues at CICIMAR: Dr. Fransisco Arreguín-Sánchez, Dr. Manuel Jesus Zetina Rejon, Dr. Victor H. Cruz-Escalona and

Mirtha Albañez Lucero; and at ENIP: Gerónimo Rodríguez Chávez, Rodolfo Navarro Villalobos, Jose Trinidad Nieto Navarro, and Delia Ojeda. In addition I would like to thank Dr. Felipe Amezcua, Enrique Morales-Bojorquez and the many other individuals who made time to meet with me for important conversations that helped shape my research.

A tremendous amount of data collection is behind the chapters of this thesis, and I would still be dissecting fish and entering data were it not for the unwavering dedication of my research assistants. Many thanks to Alex Fung, Ivan Martínez-Tovar, Jesús Antonio Cárdenas, Enrique Gallegos, Jesús Ernesto Rincones, Pui Yee Lau, Sergio Paúl Padilla, and Sophia Sandoval. You proved to be some of the most capable and hard working people I have had the pleasure to meet. Thank you also to the technicians who helped collect my fish samples onboard the boats: Armando Hernández and Daniel Cisneros.

In addition to helping with important logistics, my Mexican colleagues provided me with an incredible amount of friendship and support. You each went out of your way to ensure my field work was successful on both professional and personal levels, and I am deeply thankful for your support and friendship (and your patience as I improved my Spanish!). In particular, I owe a great deal of my accomplishments to the overwhelming hospitality of Trini, Delia and Norman. Thank you for opening up your home to me – you are special members of my family.

There are not words to thank the fishers who generously shared their fishing vessels with me for an entire shrimp season: Bam Bam, Becerro, Chile, Chorejas, Cirol, Cocha recio, Cuate, Gil, Nicho, Paisa, Paquita, Pariente, Punzada, Toño, Ulyses, and Valdo. Thank you for inviting me onto your boats and into your families. I know it was not easy for you to have me onboard – but we learned to be a good team. By opening up your hearts and minds, you taught me more than any research paper could about the realities of your livelihood. You also kept me safe, something my mother thanks you for! You made my PhD one of the most wonderful and enriching experiences of my life. Gracias amigos.

In addition to my committee I received a lot of support and guidance at UBC and beyond. Thank you to Shayne E. MacLellan and Susan Mahannah at the Fish Ageing Lab of the Pacific Biological Station, Nanaimo, for their assistance with ageing my otoliths; to Nathan Taylor for early guidance with analyses; to Janelle Curtis and Dave O’Brien for much needed hand holding as I learned to code in R (I can not thank you enough); and to the following friends and colleagues who kindly gave their time to help me with analyses, figures, and to read draft versions of my work: Janna Rist, Jennifer Selgrath, Kerrie O'Donnell, Mai Yasue, Natalie Ban, Phillip Molloy, Regina Bestbier, Sarah Bartnik and Tiphaine Jeanniard du Dot. Extra thanks to members of the "We Will Finish Group" without whose support and friendship I would not have met my self imposed deadlines – Iain Caldwell, Jenny Selgrath and Louise Blight.

One of the best parts of being a Project Seahorse team member is getting to work alongside many fun and dedicated people. In addition to those already named, thanks to the following individuals for their logistic and moral support: Amado Blanco, Bob Hunt, Chloe Shen, Danika Kleiber, Eulalio Guieb III, Eve Robinson, Geoff Gilliard, Heather Koldewey, Heidi Shuter, James Hrynyshyn, Jean Marcus, Jonathan Anticamara, Jorma Neuvonen, Justin Smallbridge, Keith Martin-Smith, Lana Gunnlaugson, Lucy Woodall, Marivic Pajaro, Marjorie Sorensen, Nick Hill, Nishan Perera, Sara Lourie, Shannon Charney, Sian Morgan, Sue Mulligan, Fatima Damji, and last but certainly not least, Melissa Grey.

Balance is the key to a happy life, and it is my amazing family and friends from coast to coast, and around the world, who provided me much needed balance on this topsy-turvy journey. I will not name all of you here (for fear of forgetting someone!), but you know who you are. Thank you for always believing in me, even when I did not believe. Thank you for being interested in what I do, and reminding me that it is interesting when I happened to forget. Thank you for helping me celebrate my scholarly victories, and reminding me that there are other things to celebrate when school wasn't one of them. Thank you most of all for your unconditional love. I am a better person for knowing you. A special thank you to my good friend Renee. You tragically left me at the start of this journey – or at least I thought you had. But then I found you when afloat on a small shrimp boat, and realised you never left at all. I love you.

Mom and Dad – it is your unwavering belief in me that has, and always will, keep me going even when I want to stop. You are responsible for opening my eyes and heart to the amazing gifts nature has to offer, and for instilling in me a deep sense of responsibility to protect them – one "starfish" at a time. Mom, if we all practiced your "take one for every ten" rule the world would be a better place. And Dad, thanks for teaching me how to gut a fish – who knew that skill would come in so handy! I love you both to the end of the numbers.

Last, but definitely not least, I owe the world to my husband, Marc Raymond (aka Poncho). You have been my constant partner on this journey, and I could not have asked for more loving, supportive, and fun company. This thesis is every bit as much your accomplishment as it is mine. I love you.

To my husband, Marc,

and to my parents, Bill and Joan

– Your love made this journey possible

"It is good to have an end to journey toward;

but it is the journey that matters, in the end."

Ursula K. Le Guin

Co-authorship statement

With the exception of Chapters 1 and 6, all chapters have been prepared as stand-alone, peer-reviewed publications. Chapters 2 to 4 are presently at refereed journals, while Chapter 5 in is preparation for submission. I am the senior author on all papers. I am the sole author on Chapter 3, although Dr. Amanda Vincent and Dr. Scott Hinch provided excellent guidance. Dr. Amanda Vincent is a co-author for all remaining manuscripts. For each of the co-authored manuscripts I collected all information for the paper, carried out all data management and analyses, drafted the text, and created all figures and tables. Dr. Amanda Vincent was instrumental in developing the thesis framing, played a key role in developing its logistic and intellectual trajectory, and then provided edits, comments and suggestions for improvement on drafts of each chapter.

1. Introductory chapter

Rationale

The goal of this thesis is to find out whether tropical shrimp trawling is a problem for the viability of the thousands of incidentally captured small fish species. The research was prompted when a significant conservation advance revealed a critical gap in our understanding of threats facing small fishes in the world's tropical seas. All species of seahorse (genus *Hippocampus*) were listed on CITES Appendix II in 2002 (Foster & Vincent 2005). As a consequence, member countries wishing to export seahorses must prove such exports are not detrimental to survival of their wild populations. Yet the majority of seahorses entering international trade are caught incidentally by the world's tropical shrimp trawlers (Vincent 1996; Baum & Vincent 2005; Giles et al. 2006), and nothing is currently known about the impacts of such indiscriminate fishing practices on any seahorse population. Since seahorses are captured with hundreds of other small fish species, we had hoped to infer impacts on seahorses from information on the effects of shrimp trawling on other small fishes. However, at the time that seahorses were listed on CITES, a review of the literature revealed that while progress has been made in understanding the impacts of some nonselective fishing practices on several bycatch taxa (S.J.F. unpublished data), there remain large gaps in our understanding of shrimp trawling on the majority of the species in the bottom of the nets – the small fishes. This thesis is a first attempt to address this gap.

Background

Fish are important sources of food and income, and many have recreational and cultural value, but most of the world's major fisheries are overfished or on the edge of collapse (Delgado et al. 2003; Garcia & Grainger 2005). Indiscriminate fishing practices worsen the problem, by wasting a high percentage of each year's catch; many millions of tonnes of non-target species (bycatch) are swept up and discarded by commercial fishing operations each year (Alverson et al. 1994; Kelleher 2004). Increasing concern about the effects of nonselective fishing practices on non-target species has led to the inclusion of calls for bycatch management in various international agreements/treaties. For example, both the United Nations Convention on the Law of the Sea (UN 1982), and the Food and Agriculture Organisation Code of Conduct for

Responsible Fisheries (FAO 1995), require that fishing be conducted in a manner which does not undermine the sustainability of bycatch species.

Of all non-selective fisheries, tropical shrimp trawlers generate the highest rate of discards, accounting for approximately 30% of the total global estimate (Alverson et al. 1994; Kelleher 2004). Such wasteful fishing, where it goes unmanaged, has the potential to deplete populations, degrade habitats, damage ecosystems and diminish food security. The majority of the bycatch species are small fishes, those maturing at less than 20 cm and weighing less than 100 grams (Alverson et al. 1994; Kelleher 2004). These incidentally caught fishes are in the same size range as the target shrimps. The small fish component of the catch is often labelled as 'trash', even in the scientific literature, leaving no question as to the low value in which they are held. The majority of 'trash fish' are simply discarded to sea, but in some countries they are kept to be turned into fishmeal and fish oil for agriculture and aquaculture operations (Clucas 1997).

The majority of small fish species obtained as bycatch by industrial tropical shrimp trawl fisheries have low economic value (Gillet 2008), but some may have considerable ecological value, and many may become more valuable to humans as time progresses. Where small fish species play important roles as prey, competitors, predators, and herbivores, they are key determinants of how other populations fare. We need to increase our understanding of the ecological role played by incidentally captured small fishes. Meanwhile, common sense requires that we keep these building blocks in place if ecosystems are to function effectively. Small fishes are also gaining importance in supplying food to people. As human populations continue to grow, demands for fish meal for aquaculture increase (Naylor et al. 2000; Delgado et al. 2003), and we continue to "fish down the marine food web" (Pauly et al. 1998). These increasing pressures mean small fish may become considerably more important than they are now for food security, making it crucial that populations are maintained at sustainable levels. Virtually nothing is known about impacts of tropical shrimp trawls on the thousands of small fish they catch; this is cause for concern. Commitments to ecosystem-based management (e.g. Sherman et al. 2005), and future food supply (e.g. Delgado et al. 2003), mean we must find ways to assess and address trawler impacts on small fishes.

Potential impacts

General descriptions of bycatch are common, but the study of impacts has hitherto been reserved for developed countries, megafauna, and valuable fishes. Most research on understanding impacts of nonselective fishing practices has been carried out in temperate systems where sufficient resources exist to conduct experimental trawling and long-term monitoring, such as the North Sea and the North Atlantic (e.g. see refs in Kaiser & Groot 2000). Similarly, the majority of research into the impacts of tropical shrimp trawl fisheries has occurred in developed nations with sufficient capacity to undertake such studies – Australia and the US Gulf of Mexico (e.g. Harris & Poiner 1990; Hendrickson & Griffin 1993; Brewer et al. 1998; Ortiz 1998; Robins et al. 1999; Stobutzki et al. 2001b). This is an important geographical bias given that the majority of tropical shrimp trawl fisheries are found in developing countries (Andrew & Pepperell 1992; Gillet 2008). The concentration of shrimp trawl fisheries in developing countries means managers will place a premium on cost effective and pragmatic methods of assessment and action, which benefit fish and fishers alike.

Research into population level impacts of tropical shrimp trawling on bycatch species is scarce. The focus of the few existing studies has been on charismatic megafauna such as turtles (e.g. Caillouet et al. 1996; Poiner & Harris 1996; McDaniel et al. 2000; Lewison et al. 2003), commercially important fish species (e.g. Gallaway et al. 1997; Ortiz 1998; Diamond et al. 1999), and the benthos (e.g. Burridge et al. 2003). Impact assessments have been executed for only a few commercially important large species such as red snapper (*Lutjanus campechanus*), weakfish (*Cynoscion regalis*), and Atlantic croaker (*Micropoganias undulatus*) (Crowder & Murawski 1998). The attention paid to these higher-profile and sensitive bycatch species has increased public awareness of the issue (Horsten & Kirkegaard 2002). It is time, however, to consider the many hundreds of small fish species about which we know virtually nothing.

Non-selective fishing practices may affect small fishes through (a) direct mortality, (b) indirect mortality, (c) community disturbance, and (d) habitat damage (Andrew & Pepperell 1992). Fishing mortality can be attributed to direct removal, including that which occurs after discarding (and post-discarding mortality rates for small fishes are high, Hill & Wassenberg 1990; Sangster et al. 1996; Wassenberg et al. 2001). Fishing mortality can also be indirect, where fish die due to damage sustained from escaping the gear, or where their injuries increase their susceptibility to predation (e.g. Sangster et al. 1996; Ryer 2002). Such indirect sources of fishing mortality are hard to monitor and generally go unnoticed (Horsten & Kirkegaard 2002). Small fish populations may also be affected where fishing practices shift the balance of the ecosystem. For example, decreases in predators may allow small fish populations to increase, and trawling induced changes to benthic communities and/or food subsidies from discarding practices may variably affect the diet of small fishes (Jennings & Kaiser 1998; Fonds & Groenewold 2000; Greenstreet & Rogers 2000). Finally, bottom trawls can cause significant amounts of change to important fish habitats (e.g. Kaiser 1998; Thrush et al. 1998; Kaiser 2000; Kaiser et al. 2000; Lindegarth et al. 2000; Piet et al. 2000; Kaiser et al. 2002; National Research Council 2002; Thrush & Dayton 2002; Burridge et al. 2003). As an example, a loss of biogenic structures may increase the predation risk for small species (Jennings & Kaiser 1998).

The diverse ways in which nonselective fishing practices affect small fishes probably explains their variable responses to trawling. In some cases populations have declined – populations of ponyfish (Leiognathidae) were observed to decrease in both Malaysia and the Gulf of Thailand after the introduction of shrimp trawling (Pauly & Neal 1985; Chan & Liew 1986). But while fishing increases mortality for many non-target species, other species may benefit such that their populations increase in numbers in response to trawling. This appeared to be the case for saurids in the Gulf of Thailand (Longhurst & Pauly 1987). Similar observations have been made in temperate systems. In the North Sea, trawling increased population size of gobies by removing large predators, and short-term increases in densities of whiting were observed as they scavenge on benthic species killed by the trawl (Fonds & Groenewold 2000).

Assessing impacts

Research into bycatch from shrimp trawl fisheries has, to date, mostly focused on discard ratios as well as raw numbers and weight, neither of which are necessarily indicators of adverse biological or ecological consequences (Alverson & Hughes 1996). Indeed, the few studies that have suggested effects of shrimp trawling on small fish species have been based on observed changes in abundance (as outlined above). When it comes to determining actual impact of nonselective fishing practices on bycatch species, most researchers have used analyses of time series of either fisheries-dependent or independent catch, effort and/or abundance data (Ortiz 1998; Diamond et al. 1999; Bergman & van Santbrink 2000; Diamond et al. 2000; Pope et al. 2000). It is unlikely that such long-term data exist for small fish species obtained as bycatch in tropical shrimp trawl fisheries. How then are we to quantitatively determine the impacts of tropical shrimp trawl fisheries on small fish species in bycatch?

Even with few data, one should be able to deduce the potential consequences of indiscriminate fishing from an understanding of the bycatch species' life history, ecology, and population parameters (Dulvy et al. 2004). Life history rates (survival, growth, reproduction, movement) of marine fishes have important implications for their population-level responses to exploitation and habitat loss. Theoretical population dynamics suggest that populations of slow-growing, large bodied and late maturing species should decrease more quickly than populations of fast-growing, relatively small, and early maturing animals, when subject to similar mortality rates (e.g. Jennings et al. 1998; Jennings et al. 1999a; Jennings et al. 1999b). Small fish species generally fall into the latter category, but behavioural characteristics may also be important in determining susceptibility such as territoriality, mobility, mating patterns and parental care (Stobutzki et al. 2001a; Dulvy et al. 2003).

Correlative assessments and demographic approaches using life history parameters may give insight into which small fish species are more vulnerable to bycatch mortality (Dulvy et al. 2004). Assessment methods might include, for example, length-based indicators, qualitative criteria, and population viability analysis. Froese (2004) suggested three simple length-based indicators to assess a population for overfishing: percent of retained fish that are (a) mature, (b) at optimum length, and (c) mega-spawners. This simple assessment requires, at a minimum, the length frequency distribution of the catch and an estimate of length at maturity. Stobutuzki et al. (2001a) proposed assessing bycatch species based on their susceptibility to capture and capacity for recovery. Susceptibility to capture was inferred from a species' ecology, while recovery capacity was determined from life history and population characteristics. In such a framing, small did not equal safe; the fishes least likely to be extracted sustainably were mostly small demersal species with somewhat specialised life histories, whereas larger pelagic species were most likely to be sustainable. Stage-based population viability models can be constructed with information on length frequency distributions, length at maturity, fecundity at length, sex ratios, growth parameters, and maximum age (Braccini et al. 2006).

Socioeconomic realities

Tropical shrimp trawl fisheries may have strong impacts on the environment, but they are perceived as important to the countries in which they are based. Shrimp is considered the most valuable fishery export for many tropical developing countries, with the fisheries employing a significant number of people (Gillett 2008). However, the majority of large-scale (industrial) tropical shrimp trawl fisheries are economically overfished (Gillett 2008). Too many boats, with ever larger gears, have led to global declines in shrimp catch per unit effort, and in many places it is thought that tropical shrimp resources are now overfished (Gillett 2008). Declining catch rates combined with rising overhead costs (mainly fuel) and falling shrimp prices (due to world wide competition with lower-cost farmed shrimp), have led to low profitability for most of the worlds commercial shrimp fishing operations (Gillett 2008). In many cases government-funded subsidies have been required for continued operation (Gillett 2008). Indeed, half of all shrimp landings presently come from countries with subsidies on fuel (Sumaila et al. 2008).

Any proposed mitigation tools must respect the socioeconomic context of the fisheries and the region so as to engender compliance, or at least to be enforceable. Industrial tropical shrimp trawl fisheries may eventually become so unprofitable that they cease to exist, solving the bycatch issue without much intervention. In the meantime, however, their purported importance means that stakeholders (i.e. governing bodies, fishers) will require incentives to decrease their impacts, where they exist, on small fishes that have less economic value than the targeted shrimp. Policy and plans to protect nature are only successful if they are accepted and supported by key stakeholders (Leslie 2005). Interdisciplinary approaches are thus needed to address the issue of tropical shrimp trawl bycatch. There are, luckily, some broad benefits in reducing bycatch of small fishes: reduce detrimental effects on commercially important species that may be affected by removing components of their ecosystem; avoid killing rare or protected animals; reduce sorting times; avoid rotting materials which contaminate catches; and avoid criticisms of the fishery due to waste (Andrew & Pepperell 1992).

Addressing impacts

Fisheries management for sustainability includes regulating the quantity of fish caught, when and where they are caught, and the size at which they are caught (Cochrane 2002). This is true for all catch, whether target or incidental. Managers can achieve these goals for bycatch species by reducing overall effort, which may also reduce target catch, or by reducing bycatch per unit effort with potentially less impact on the fishery (Hall 1995). The latter requires increasing the fisheries selectivity, either through gear specifications (e.g. mesh regulations or bycatch reduction devices) or in time and/or space (e.g. introducing time/area closures in bycatch hotspots). In some cases bycatch and/or discard quotas are used, where fisheries can be closed when the quota, either for specific species or aggregated bycatch, is met (Horsten & Kirkegaard 2002). Many shrimp trawl fisheries are too big, however, for individual species bycatch quotas to be practical (Diamond 2004). Incentive-based programs have also been proposed, such as increasing total catch limits for fishers with low bycatch to target catch ratios (Horsten & Kirkegaard 2002).

Reducing bycatch through the development of gear modifications is the solution that has received the most attention (e.g. Broadhurst 2000; Kennelly & Broadhurst 2002; Eayrs 2007). As with understanding impacts, the majority of work on technological changes for reducing bycatch has occurred in temperate regions. In temperate systems, development of gear and techniques for increasing fishery selectivity is feasible because of the relatively few bycatch species, distinct difference in size between shrimp and fish species, and passive behaviour of targeted shrimps (Broadhurst 2000). The situation is not the same for tropical shrimp fisheries, where the bycatch is highly diverse, and the majority of the bycatch species overlap in size with targeted shrimps (Brewer et al. 1998; Cochrane 2002). As a result, selective gear and techniques used in higher latitudes are not readily transferable to tropical waters (Clucas 1997). Indeed, it may never be possible to find a technological solution that works for all small fishes, given their vast array of behaviour patterns, physiological condition, body form, and morphology (Bublitz 1995; Loverich 1995). Further, even where alternative technologies might be available for tropical fisheries, cost can prohibit their implementation (Gillett 2008).

For small fish species the available solutions that are likely to work best are time/area closures to trawling. Where the bycatch problem is not easily solved with improved gear selectivity, reducing bycatch becomes dependent on the ability to target fishing in time and space (Horsten & Kirkegaard 2002). Protected areas can be useful for mitigating problems on a variety of fishes with divergent life history strategies, whereas other management strategies that are designed to help one species may have unanticipated effects on sympatric species that exhibit alternative strategies (Winemiller & Rose 1992). Trawling restrictions reduce effort on populations, but should also reduce ecosystem disruption and habitat damage, thereby addressing all possible types of impacts on bycatch species. Implementing closures during times when small fish species are abundant or most vulnerable may reduce bycatch without huge implications for the shrimp fishery (Andrew & Pepperell 1992). There are few studies demonstrating the effects of trawl closures for bycatch mitigation. A seasonal closure in Kuwait, originally implemented to increase the size of shrimps before capture, and the closure of Kuwait Bay to trawling, both appear to have reduced bycatch (Ye et al. 2000).

Greater retention of bycatch species is frequently listed as a mitigation measure for tropical shrimp trawl fisheries (Hall 1995; Clucas 1997; Hall & Mainprize 2005). In some cases it is hoped that the limited holding space and increased costs of retaining fishes of little value would provide incentive to reduce what is caught in first place (Halley & Stergiou 2005). In most cases, however, the literature calls for retention of the bycatch so that it can be used instead of wasted at sea (Andrew & Pepperell 1992). To this end new markets are being established for small fish bycatch. Such practices may be positive where the bycatch is efficiently used to increase human food security, but instead the majority of small fish are inefficiently converted into feed for aquaculture and agriculture (Gillet 2008). In such cases there is a loss of fish for human consumption because of the low conversion rate of fish to human food, much of which is consumed in developed countries instead of the developing nations from where the fish originated (Gillet 2008). In addition, and critically, retaining the bycatch does not mitigate environmental impact (Davies et al. 2009). Indeed, it can worsen it, in that the bycatch becomes a secondary catch and can help drive or subsidise otherwise failing shrimp trawling (Gillet 2008; Lobo et al. 2009). We need to reduce the bycatch that is obtained in the first place.

Research objectives

My research is framed within three main objectives. The first objective was to use newly derived life history information for several small fish species to evaluate possible effects of their incidental capture. The second objective was to shed light on the social dimensions of tropical shrimp fisheries management. The third objective was to consider the potential value of socially acceptable bycatch mitigation measures for small fishes.

To meet these objectives I asked the following questions with my research:

1. What are the life history and population parameters of small fish species obtained as bycatch in a tropical shrimp trawl fishery? (First objective)
2. What do the life history and population parameters tell us about potential risk for small fish species from the tropical shrimp trawl fishery? (First objective)
3. What are industrial shrimp fishers' views on direct and indirect problems facing their fishery, proposed and potential management options to address the issues, and the future of the fishery in general? (Second objective)
4. What are socially accepted and biologically appropriate bycatch mitigation measures that may be used to address the potential risk conferred on these species from their incidental capture in the fishery? (Third objective)

Context and collaborations

This thesis is focused on the industrial shrimp trawl fishery in the Gulf of California, Mexico, a useful model for tropical shrimp trawl fisheries. Although only the southernmost portion of the Gulf is within the latitudinal boundaries of the inter-tropical zone, the Gulf is considered to be part of the Eastern Tropical Pacific. In addition, its shrimp fishery has more in common with those in tropical rather than temperate zones (e.g. Gillett 2008). Understanding the impacts of the Pacific shrimp trawl fishery in the Gulf of California is a pressing issue. The Gulf of California has become the most important body of water in Mexico for fisheries, providing more than half of all Mexican landings by volume (Arvizu-Martinez 1987), and contains the country's main shrimp trawling grounds (Magallon-Barajas 1987). The Gulf's shrimp trawl fishery is considered to be of great social and economic importance, and is credited with most of the economic growth in the surrounding states (García-Caudillo & Gómez-Palafóx 2005). The Gulf of California is also one of the world's most productive and biologically diverse marine ecosystems (Enríquez-Andrade et al. 2005). For example, approximately 4800 invertebrate species (of which more than 740 are endemic), 875 fish species (of which 77 are endemic), five the world's seven sea turtle species, and almost 40% of known cetacean species use the Gulf to feed and breed (various refs in Enríquez-Andrade et al. 2005).

In spite of the declared socio-economic and ecological importance of the Gulf of California, there are few quantitative assessments of the impact of the shrimp fishery on a species or ecosystem level (Morales-Zárate et al. 2004). Past research indicated that the incidental catch of the shrimp fishery in the Gulf of California included over 100 species of fish, and 92% of the bycatch consisted of fish individually weighing about 50 grams and measuring 10-20 cm in total length (Perez-Mellado & Findley 1985). The average shrimp to bycatch ratio by weight was estimated at 1:13 (Perez-Mellado & Findley 1985). This information, while providing a point of comparison, needs updating, and none of it demonstrates an impact of the fishery on small fish species.

An overview of Mexico's industrial shrimp fishery can be found in Gillet 2008. Typical shrimp trawlers from the Gulf of California are double otter trawls, such that one vessel tows two otter trawls from the ends of outrigger booms. Each trawl is a cone-shaped net closed by a cod-end with diamond shaped meshes, held open by two otter boards made of wood. A detailed description of the gear, along with diagrams, can be found in Gillet 2008 (pages 17 – 20).

To carry out an interdisciplinary approach to solving shrimp trawl bycatch for small fish species, I developed partnerships with a team at the Centro Interdisciplinario de Ciencias Marinas (CICIMAR) in La Paz, Baja California. There, Dr. Arreguín-Sánchez and his group have been studying the effects of shrimp trawling on marine environments, evaluating the bycatch statistics against current knowledge on trophic levels and connections among species. My research on small fishes will complement their research and be valuable for refining their ecosystem-based models. Through this connection, I also worked with Escuela Nacional de

Ingeneria Pesquera in San Blas, Nayarit, and Instituto de Ciencias del Mar Y Limnologia in Mazatlán, Sinaloa.

By collaborating with one of CICIMAR's ongoing projects, I had the amazing opportunity of being able to participate in fishing trips onboard two industrial shrimp trawlers in the southern Gulf of California, Ing. Salvador Villaseñor A. and Ing. Miguel Lopez Rivera. Dr. Arreguín-Sánchez and his team at CICIMAR commission these boats to collect data for their study on the effects of shrimp trawling on the Gulf's ecosystem. Because of their arrangement, I was able to sample the bycatch each month, for one entire fishing season (September 2006 – March 2007), from 38 fixed stations along the coast of the southern Gulf of California. Such systematic sampling is rarely afforded through fisheries dependent research, and greatly contributed to the strength of my results.

Thesis outline

This thesis has four data-based research chapters, followed by a general synthesis. Chapters 2, 3 and 5 address the first and third objectives of my thesis, while Chapter 4 addresses the second objective.

In Chapter 2 I characterise, through fisheries dependent sampling, the life history of four small fish taxa obtained as bycatch in the southern Gulf's shrimp trawl fishery, and evaluate the potential for impact on these species based on their life histories. The following four taxa were chosen as case studies for this research: sand perch (*Diplectrum* spp.), lumptail searobin (*Prionotus stephanophrys*), bigscale goatfish (*Pseudupeneus grandisquamis*) and silver stardrum (*Stellifer illecebrosus*).

In Chapter 3, I further assess potential for impacts from trawling on two of the small fish species identified as potentially vulnerable in Chapter 2, bigscale goatfish and silver stardrum. In this chapter I use their distributions in space and time in the southern Gulf to assess impact and evaluate the potential use of spatio-temporal trawl restrictions to mitigate impact for these species.

In Chapter 4, I use interviews with industrial shrimp trawlers from the two main ports in the southern Gulf to increase our understanding of socio-economic realities of managing shrimp fisheries for sustainability (objective 2). The results from this chapter were used to identify socially acceptable bycatch mitigation measures for consideration in other chapters (objective 3).

The results of Chapters 2 and 3 identified one particular species for which the Gulf's industrial shrimp trawl fishery may be a problem, silver stardrum. In Chapter 5, I integrate the life history information for this species from previous chapters (2 and 3) into a deterministic matrix analysis. I used the matrix model to determine the population's status and explore the potential of fisher supported bycatch mitigation measures, identified in Chapter 4, to address impact.

Finally, I end with a general discussion of the findings presented in this thesis and how they contributed to meeting my research objectives (Chapter 6).

Literature cited

Alverson, D. L., M. H. Freeberg, J. G. Pope, and S. A. Murawski. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper No. 339: 1-233.

Alverson, D. L., and S. E. Hughes. 1996. Bycatch: from emotion to effective resource management. Reviews in Fish Biology and Fisheries **6**: 443-462.

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

Arvizu-Martinez, J. 1987. Fisheries activities in the Gulf of California, Mexico. CalCOFI Report **XXVIII**: 32-36.

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

Bergman, M. J. N., and J. W. van Santbrink. 2000. Fishing mortality of populations of megafauna in sandy sediments. Pages 49-68 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Braccini, J. M., B. M. Gillanders, and T. I. Walker. 2006. Hierarchical approach to the assessment of fishing effects on non-target condricthyans: case study of *Squalus megalops* in southeastern Australia. Canadian Journal of Fisheries and Aquatic Sciences **62**: 2456-2466.

Brewer, D., N. Rawlinson, S. Eayrs, and C. Burridge. 1998. An assessment of bycatch reduction devices in a tropical Australian prawn trawl fishery. Fisheries Research **36**: 195-215.

Broadhurst, M. K. 2000. Modifications to reduce bycatch in prawn trawls: A review and framework for development. Reviews in Fish Biology and Fisheries **10**: 27-60.

Bublitz, C. G. 1995. Mesh size and shape: Reducing the capture of undersized fish. Pages 95-99. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Burridge, C. Y., T. J. Pitcher, T. J. Wassenberg, I. R. Poiner, and B. J. Hill. 2003. Measurement of the rate of depletion of benthic fauna by prawn (shrimp) otter trawls: an experiment in the Great Barrier Reef, Australia. Fisheries Research **60**: 237-253.

Caillouet, C. W., D. J. Shaver, W. G. Teas, J. M. Nance, D. B. Revera, and A. C. Cannon. 1996. Relationship between sea turtle stranding rates and shrimp fishing intensities in the northwestern Gulf of Mexico: 1986-1989 versus 1990-1993. Fishery Bulletin **94**: 237-249.

Chan, E. H., and H. C. Liew. 1986. Characteristics of an exploited tropical shallow-water demersal fish community in Malaysia. Pages 349-352 in J. L. Maclean, L. B. Dizon, and

L. V. Hosillos, editors. The First Asian Fisheries Forum. Asian Fisheries Society, Manila.

Clucas, I. 1997. A study of the options for the utilization of bycatch and discards from marine capture fisheries. Page 59. FAO Fisheries Circular No. 928. Food and Agriculture Organisation of the United Nations, Rome.

Cochrane, K. L., editor. 2002. A fishery manager's guidebook - Management measures and their application. Food and Agriculture Organisation of the United Nations (FAO), Rome.

Crowder, L. B., and S. A. Murawski. 1998. Fisheries bycatch: implications for management. Fisheries **23**: 8-17.

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

Delgado, C. L., N. Wada, M. W. Rosegrant, S. Meijer, and M. Ahmed 2003. Outlook for fish to 2020: meeting global demand. International Food Policy Research Institute and World Fish Centre, Washington, Penang.

Diamond, S. L. 2004. Bycatch quotas in the Gulf of Mexico shrimp trawl fishery: can they work? Reviews in Fish Biology and Fisheries **14**: 207-237.

Diamond, S. L., L. B. Cowell, and L. B. Crowder. 2000. Population effects of shrimp trawl bycatch on Atlantic croaker. Canadian Journal of Fisheries and Aquatic Sciences **57**: 2010-2021.

Diamond, S. L., L. B. Crowder, and L. B. Cowell. 1999. Catch and bycatch: The qualitative effects of fisheries on population vital rates of Atlantic croaker. Transactions of the American Fisheries Society **128**: 1085-1105.

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

Dulvy, N. K., Y. Sadovy, and J. D. Reynolds. 2003. Extinction vulnerability in marine populations. Fish and Fisheries **4**: 25-64.

Eayrs, S. 2007. A Guide to Bycatch Reduction in Tropical Shrimp-Trawl Fisheries. FAO, Rome.

Enríquez-Andrade, R., G. Anaya-Reyna, J. C. Barrera-Guevara, M. d. l. Á. Carvajal-Moreno, M. E. Martínez-Delgado, J. Vaca-Rodríguez, and C. Valdés-Casillas. 2005. An analysis of critical areas for biodiversity conservation in the Gulf of California region. Ocean and Coastal Management **48**: 31-50.

FAO 1995. Code of Conduct for Responsible Fisheries. Food and Agriculture Organisation, Rome.

Fonds, M., and S. Groenewold. 2000. Food subsidies generated by the beam-trawl fishery in the southern North Sea. Pages 130-150 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Foster, S. J., and A. C. J. Vincent. 2005. Enhancing sustainability of the international trade in seahorses with a single minimum size limit. Conservation Biology **19**: 1044-1050.

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

Gallaway, B. J., M. Longnecker, J. G. Cole, and R. M. Meyer. 1997. Estimates of shrimp trawl bycatch of red snapper (*Lutjanus campechanus*) in the Gulf of Mexico. Pages 817-839 in F. Funk, T. J. Quinn II, J. Heifetz, J. N. Lanelli, J. E. Powers, J. F. Schweeigert, P. J. Sullivan, and C. I. Zang, editors. Fisheries Stock Assessment Models. Alaska Sea Grant College Program.

García-Caudillo, J. M., and J. V. Gómez-Palafóx 2005. La pesca industrial de camarón en el Golfo de California: Situación económico-financiera e impactos socio-ambientales. Conservación Internacional-Región Golfo de California, Guaymas.

Garcia, S. M., and R. J. R. Grainger. 2005. Gloom and doom? The future of marine capture fisheries. Philosophical Transactions of the Royal Society of London B **360**: 21-46.

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

Gillett, R. 2008. Global study of shrimp fisheries. Food and Agriculture Organization of the United Nations, Rome.

Greenstreet, S. P. R., and S. I. Rogers. 2000. Effects of fishing on non-target fish species. Pages 217-234 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Hall, M. A. 1995. Strategic issues in managing bycatch. Pages 29-39. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

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

Halley, J. M., and K. I. Stergiou. 2005. The implications of increasing variability of fish landings. Fish and Fisheries **6**: 266-276.

Harris, A. N., and I. R. Poiner. 1990. By-catch in the prawn fishery of Torres Strait: Composition and partitioning of the discards into components that float or sink. Australian Journal of Marine and Freshwater Research **41**: 37-52.

Hendrickson, H. M., and W. L. Griffin. 1993. An analysis of management policies for reducing shrimp by-catch in the Gulf of Mexico. North American Journal of Fisheries Management **13**: 686-697.

Hill, B. J., and I. R. Wassenberg. 1990. Fate of discards from prawn trawlers in Torres Straight. Australian Journal of Marine and Freshwater Research **41**: 53-64.

Horsten, M. B., and E. Kirkegaard. 2002. Bycatch from a perspective of sustainable use. Page 16. IUCN SSC European Sustainable Use Specialist Group, Brussels.

Jennings, S., S. P. R. Greenstreet, and J. D. Reynolds. 1999a. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology **68**: 617-627.

Jennings, S., and M. J. Kaiser. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology **34**: 201-352.

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

Jennings, S., J. D. Reynolds, and N. V. C. Polunin. 1999b. Predicting the vulnerability of tropical reef fishes to exploitation with phylogenies and life histories. Conservation Biology **13**: 1466-1475.

Kaiser, M. J. 1998. Significance of bottom-fishing disturbance. Conservation Biology **12**: 1230-1235.

Kaiser, M. J. 2000. The implications of the effects of fishing on non-target species and habitats. Pages 383-392 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Kaiser, M. J., J. S. Collie, S. J. Hall, S. Jennings, and I. R. Poiner. 2002. Modification of marine benthic habitats by trawling activities: prognosis and solutions. Fish and Fisheries **3**: 114-136.

Kaiser, M. J., and S. J. Groot, editors. 2000. Effects of Fishing on Non-Target Species and Habitats. Blackwell Science, Ltd., Oxford.

Kaiser, M. J., K. Ramsay, C. A. Richardson, F. E. Spence, and A. R. Brand. 2000. Chronic fishing disturbance has changed shelf sea benthic community structure. Journal of Animal Ecology **69**: 494-503.

Kelleher, K. 2004. Discards in the world's marine fisheries: An update. FAO, Rome.

Kennelly, S. J., and M. K. Broadhurst. 2002. By-catch be gone: changes in the philosophy of fishing technology. Fish and Fisheries **3**: 340-355.

Leslie, H. M. 2005. A synthesis of marine conservation planning approaches. Conservation Biology **19**: 1701-1703.

Lewison, R. L., L. B. Crowder, and D. J. Shaver. 2003. The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the western Gulf of Mexico. Conservation Biology **17**: 1089-1097.

Lindegarth, M., D. Valentinsson, M. Hansson, and M. Ulmestrand. 2000. Interpreting large-scale experiments on effects of trawling on benthic fauna: an empirical test of the potential effects of spatial confounding in experiments without replicated control and trawled areas. Journal of Experimental Marine Biology and Ecology **245**: 155-169.

Lobo, A. S., R. Arthur, A. Manica, and A. Balmford. 2009. Chickenfeed: Fishing for scraps in an unpredictable sea. International Marine Conservation Congress. All Academic, Inc., Fairfax.

Longhurst, A. R., and D. Pauly 1987. Ecology of tropical oceans. Academic Press, San Diego.

Loverich, G. F. 1995. Thinking beyond the traditional codends. Pages 101-106. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Magallon-Barajas, F. J. 1987. The Pacific shrimp fishery of Mexico. CalCOFI Report **XXVIII**: 43-52.

McDaniel, C. J., L. B. Crowder, and J. A. Priddy. 2000. Spatial dynamics of sea turtle abundance and shrimping intensity in the US Gulf of Mexico. Conservation Ecology **4**.

Morales-Zárate, M. V., F. Arreguín-Sánchez, J. Martínez-López, and S. E. Lluch-Cota. 2004. Ecosystem trophic structure and energy flux in the Northern Gulf of California, México. Ecological Modelling **174**: 331-345.

National Research Council 2002. Effects of trawling and dredging on seafloor habitat. National Academy Press, Washington, D.C.

Naylor, R. L., R. J. Goldburg, J. H. Primavera, N. Kautsky, M. C. M. Beveridge, J. Clay, C. Folke, J. Lubchenco, H. Mooney, and M. Troell. 2000. Effect of aquaculture on world fish supplies. Nature **405**: 1017-1024.

Ortiz, M. 1998. Shrimp bycatch in the US Gulf of Mexico: Review and evaluation of bycatch effects on exploited fish stocks. PhD. University of Miami, Coral Gables, Florida.

Pauly, D., J. Dalsgaard, V. Christensen, R. Froese, and F. J. Torres. 1998. Fishing down marine food webs. Science **279**: 860-863.

Pauly, D., and R. Neal. 1985. Shrimp vs Fish in Southeast Asian: the biological, technological and social problems. Pages 487-510 in A. Yañez-Arancibia, editor. Recursos pesqueros potenciales de Mexico: la pesca acompañante del camaron. Universidad Nacional Autonoma, Mexico.

Perez-Mellado, J. L., and L. T. Findley. 1985. Evaluacion de la Ictiofauna del camarón capturado en las costas de Sonora y norte de Sinaloa, México. Pages 201-254 in A. Yañez-Arancibia, editor. Recursos Pesqueros Potenciales de México: La pesca acompañante del camarón. Universidad Nacional Autonoma, México.

Piet, G. J., A. D. Rijnsdorp, M. J. N. Bergman, J. W. van Santbrink, J. Craeymeersch, and J. Buijs. 2000. A quantitative evaluation of the impact of beam trawling on benthic fauna in the southern North Sea. ICES Journal of Marine Science **57**: 1332-1339.

Poiner, I. R., and A. N. M. Harris. 1996. Incidental capture, direct mortality and delayed mortality of sea turtles in Australia's Northern Prawn Fishery. Marine Biology **125**: 813-825.

Pope, J. G., D. S. Macdonald, N. Daan, J. D. Reynolds, and S. Jennings. 2000. Gauging the impact of fishing mortality on non-target species. ICES Journal of Marine Science **57**: 689-696.

Robins, J. B., M. J. Campbell, and J. G. McGilvray. 1999. Reducing prawn-trawl bycatch in Australia: An overview and an example from Queensland. Marine Fisheries Review **61**: 46-55.

Ryer, C. H. 2002. Trawl stress and escapee vulnerability to predation in juvenile walleye pollock: Is there an unobserved bycatch of behaviourally impaired escapees? Marine Ecology Progress Series **232**: 269-279.

Sangster, G. I., K. Lehmann, and M. Breen. 1996. Commercial fishing experiments to assess the survival of haddock and whiting after escape from four sizes of diamond mesh cod-ends. Fisheries Research **25**: 323-345.

Sherman, K., M. P. Sissenwine, V. Christensen, A. Duda, G. Hempel, C. Ibe, P. S. Levin, D. Lluch-Belda, G. Matishov, J. McGlade, M. O'Toole, S. Seitzinger, R. Serra, H. R. Skjoldal, Q. Tang, J. Thulin, V. Vanderweerd, and K. Zwanenburg. 2005. A global movement toward an ecosystem approach to management of marine resources. Marine Ecology Progress Series **300**: 241-296.

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

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

Sumaila, U. R., L. Teh, R. Watson, P. Tyedmers, and D. Pauly. 2008. Fuel price increase, subsidies, overcapacity, and resource sustainability. ICES Journal of Marine Science **65**: 832-840.

Thrush, S. F., and P. K. Dayton. 2002. Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. Annual Review of Ecology and Systematics **33**: 449-473.

Thrush, S. F., J. E. Hewitt, V. J. Cummings, P. K. Dayton, M. Cryer, S. J. Turner, G. A. Funnell, R. G. Budd, C. J. Milburn, and M. R. Wilkinson. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecological Applications **8**: 866-879.

UN 1982. United Nations Convention on the Law of the Sea. Division for Ocean Affairs and the Law of the Sea, New York.

Vincent 1996. The International Trade in Seahorses. TRAFFIC International, Cambridge, UK.

Wassenberg, T. J., D. A. Milton, and C. Y. Burridge. 2001. Survival rates of sea snakes caught by demersal trawlers in northern and eastern Australia. Biological Conservation **100**: 271-280.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life-history diversification in North-American Fishes: Implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences **49**: 2196-2218.

Ye, Y., A. H. Alsaffar, and H. M. A. Mohammed. 2000. Bycatch and discards of the Kuwait shrimp fishery. Fisheries Research **45**: 9-19.

2. Life history information indicates potential impacts of shrimp trawling on incidentally caught small fishes[1]

[1] A version of this chapter has been submitted for publication. Foster, S.J. and A.C.J. Vincent. Using life history information to assess potential impacts from shrimp trawling on small fishes.

Introduction

Non-selective fishing may have major implications for food security, biodiversity and ecosystem function. Most of the world's major fisheries are overfished or on the edge of collapse (Garcia & Grainger 2005). Indiscriminate fishing practices worsen the problem, by wasting a high percentage of each year's catch; many millions of tonnes of non-target species (bycatch) are swept up and discarded by commercial fishing operations each year (Alverson et al. 1994). Of all non-selective fisheries, shrimp trawlers generate the highest rate of discards, accounting for approximately 30% of the total global estimate (Alverson et al. 1994; Kelleher 2004). Such wasteful fishing has potential to deplete populations, degrade habitats, damage ecosystems and diminish food security.

It is vital that analyses of fisheries impacts include more comprehensive accounts of the full ecological costs of indiscriminate gear. Previous research into the impacts of bycatch by tropical shrimp trawl fisheries has generally considered large and/or commercially important species, yet the bycatch is primarily composed of small fishes, those maturing at sizes smaller than 20 cm and weighing less than 100 grams (Alverson et al. 1994; Kelleher 2004). Some small fish species may play important roles in the ecosystem (for example, as prey for larger, commercially important, species). They are also gaining importance for human consumption, both as food (especially as we fish out larger species, Pauly et al. 1998), and for other human uses (such as fish meal for aquaculture, Naylor et al. 2000).

The concentration of tropical shrimp trawl fisheries in developing countries means managers will place a premium on cost effective and pragmatic methods of assessment and action, which benefit fish and fishers alike. Conventional fisheries stock assessments, the most commonly employed method for determining species-specific fisheries impacts, are data intensive and costly (e.g. Hilborn & Walters 1991; Walters & Martell 2004). When it comes to assessing the impact of nonselective fishing practices on bycatch species, most researchers have used analyses of time series of catch, effort and/or abundance data (Ortiz 1998; Diamond et al. 1999; Bergman & van Santbrink 2000; Diamond et al. 2000; Pope et al. 2000). It is unlikely that sufficient long-term data exist for small fish species obtained as bycatch in shrimp trawl fisheries.

With fewer data, one should be able to develop correlative assessments based on knowledge of species' life history, ecology, and population parameters (Dulvy et al. 2004). Given sufficient data, demographic approaches based on stage-based schedules of vital rates can also be considered (Dulvy et al. 2004; Braccini et al. 2006). Alternative assessment methods might include, for example, length-based indicators, qualitative criteria, and population viability analysis. Froese (2004) suggested three simple length-based indicators to assess a population for overfishing: percent of retained fish that are (a) mature, (b) at optimum length, and (c) mega-spawners. This simple assessment requires, at a minimum, the length frequency distribution of the catch and an estimate of length at maturity. Stobutuzki et al (2001) proposed assessing bycatch species based on their susceptibility to capture, and the populations' capacity to recover. Susceptibility to capture was based on a species' ecology – its distribution, habitat and depth preferences, diet, and discard survival rate. Recovery capacity was determined from life history and population characteristics – size and maturity information, removal rate, reproduction, and mortality. Stage-based population viability models can be constructed with information on length frequency distributions, length at maturity, fecundity at length, sex ratios, von Bertalanffy Growth Function (VBGF) parameters, and maximum age (Braccini et al. 2006). In addition, it has been suggested that the vulnerability of unknown bycatch species can be assessed based on responses of related stocks and species (Jennings et al. 1998).

Unfortunately, for most small fish species of no commercial value, there is a dearth of knowledge of even the basic life history and population parameters (e.g. Stobutzki et al. 2001). Given that fisheries independent sampling of fishes on the scale that is needed to truly understand life history and population structure can be costly or low priority, one can turn to fisheries dependent sampling as a cost-effective option. The objectives of this study were (1) to characterise, through fisheries dependent sampling, the life history of small fish species obtained as bycatch in a tropical shrimp trawl fishery, and (2) to apply and evaluate life history based assessment methods for data-poor species. This paper will focus on elements contributing to biological productivity (survival, growth and reproduction), and touch on vulnerability to gear. A separate paper will focus on factors related to spatial distribution, in numbers and size.

This study focused on the shrimp trawl fishery in the southern Gulf of California, Mexico. Understanding the impacts of this fishery is a pressing issue. The Gulf of California has become the most important body of water in Mexico for fisheries (Arvizu-Martinez 1987), and is also one of the world's most productive and biologically diverse marine ecosystems (Enríquez-Andrade et al. 2005). Yet few published quantitative assessments consider the impact of the shrimp fishery on a species or ecosystem level. With average bycatch to shrimp ratios of 10 to 1, and suffering from overcapacity, poor governance and perverse subsidies (García-Caudillo & Gómez-Palafóx 2005), the Gulf's shrimp trawl fishery is representative of most tropical shrimp fisheries.

The following four taxa were chosen as case studies for this research: sand perch (*Diplectrum* spp., Serranidae), lumptail searobin (*Prionotus stephanophrys,* Triglidae), bigscale goatfish (*Pseudupeneus grandisquamis*, Mullidae) and silver stardrum (*Stellifer illecebrosus,* Sciaenidae). All are small (estimated size at maturity less than 20 cm), are associated with benthic habitats (sandy and muddy bottoms; Eschmeyer et al. 1983; Chao 1995; Heemstra 1995; Schneider 1995), have potentially high abundance (to ensure sufficient sample sizes) and individuals of all life stages (juveniles to adults) were thought to be found on the shrimp trawl grounds (V. Cruz, CICIMAR, pers. comm..). These taxa have the potential to become important for food in Mexico, as most are consumed by humans elsewhere, and even form part of directed fisheries (e.g. Mendoza López 2000). Each of the focal taxa belong to one of the more commonly caught families in fishery: Serranidae – 8 spp.; Triglidae – 4 spp.; Mullidae – 2 spp.; Sciaenidae – 20 spp. (compiled from Manjarrez Acosta 2001 and CICIMAR, unpublished data).

Methods

Study species

Sandperch

At the time of field work, collected sandperch were believed to consist of a single species, highfin sandperch (*Diplectrum labarum*) (Figure 2.1a). All data were thus recorded under that name. However, a recent taxonomic revision of the composition of the bycatch from the southern Gulf, based on morphometrics, suggested that the samples contained both highfin sandperch and inshore sandperch (*Diplectrum pacificum*), and perhaps a few individuals of other conspecifics (J. Nieto, CICIMAR, pers. comm.). Unfortunately, it was not possible to go back and separate the data by species once the error had been realised.

Highfin and inshore sandperch are distributed along the Eastern Pacific Ocean, from the Gulf of California and southern Baja California, Mexico, to Panama, where they inhabit sandy and muddy bottoms (Heemstra 1995). Other sand perch species are known to maintain home ranges (Bortone 1971).

Individuals of the genus *Diplectrum* are synchronous hermaphrodites, possessing both male and female reproductive organs. Sand seabass (*Diplectrum formosum*), in the Gulf of Mexico, cross-fertilise with no possibility for self-fertilisation (Bortone 1971). In sand seabass, testes mature before the ovaries (Bortone 1971). It is unclear whether individuals are reproductively active prior to maturation of both gonads.

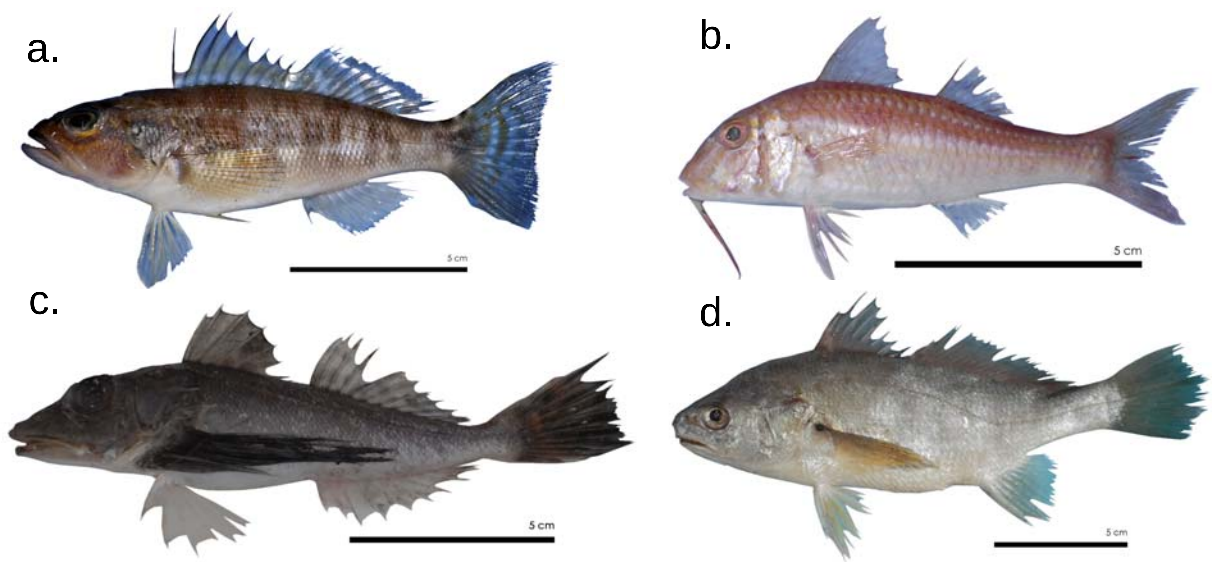

Figure 2.1. a) highfin sandperch, b) bigscale goatfish, c) lumptail searobin, and d) silver stardrum from the southern Gulf of California, Mexico. © CICIMAR.

Bigscale goatfish

Bigscale goatfish (Figure 2.1b) are distributed along the Eastern Pacific Ocean from Baja California, Mexico, to Chile, where they inhabit sandy and muddy bottoms (Schneider 1995).

The reproduction of bigscale goatfish has been studied in the Gulf of Tehuantepec, Mexico (Ramos-Santiago et al. 2006). There, reproduction occurred during all of the months studied (March, May, August, October and November), with peaks in August and October. Presence of juveniles throughout the year suggested a protracted spawning period. The total female to male sex ratio was nearly 1:1 throughout the year (Ramos-Santiago et al. 2006). Oocyte development is thought to be asynchronous within a female (Lucano-Ramírez et al. 2006).

Lumptial searobin

Lumptail searobin (Figure 2.1c) are distributed along the Eastern Pacific Ocean from the Columbia River in Washington, USA, to Chile (but are rare north of Baja California, Mexico), where they inhabit sandy and muddy bottoms (Eschmeyer et al. 1983).

In Mexico, studies of lumptail searobin have centred on quantifying growth patterns. Results have been variable, with VBGF k values ranging from 0.05-0.82 per year, and asymptotic length (L_∞) values from 243-573 mm (Schmitter-Soto & Castro-Aguirre 1991, 1994; Mendoza López 2000). The biomass of lumptail searobin in the Gulf of California may be sufficient to support a commercial fishery (Schmitter-Soto & Castro-Aguirre 1994).

Lumptail searobin has been studied in Peru, where it is of commercial importance (Castillo-Rojas et al. 2000; Samamé & Fernández 2000). There, reproduction occurs from December to March (summer), while June to September is a period of reproductive rest. Length at 50% maturity was estimated at 202 mm, while all individuals had spawned at least once by 270 mm. Maximum age was six years, with fastest growth rates between two and three years of age. Another study cited maximum age at nine years, but only a few individuals were found to be greater than six years of age (Schmitter-Soto & Castro-Aguirre 1991).

Silver stardrum

Silver stardrum (Figure 2.1d) are distributed along the coast of the Eastern Pacific Ocean from southern Gulf of California, Mexico to Peru, where they inhabit muddy bottoms (Chao 1995). No other studies have been published, and perhaps conducted, on this species.

Data collection

This research focused on the Gulf of California's southernmost shrimp fishing grounds, off the Mexican states of Sinaloa and Nayarit, from 21.25 to 23.20°N (Figure 2.2). Samples were collected from the catch of two commercial fishing vessels, at 39 fixed stations along the coast, from 25 September 2006 to 25 March 2007. The stations were spread across the shrimp trawling grounds, and were the same as those sampled annually by Mexican fisheries authorities to assess shrimp population status, and determine the opening dates for the fishery. The stations were located across an area measuring 11 600 km^2, from depths of 7 to 62 m. Samples were collected monthly, except for those along the coast of Sinaloa, where samples were not collected in November 2006 for logistical reasons. The demersal fishing gears deployed from the vessels were paired otter trawls (with 4.5-5 cm mesh size), typical of the Gulf of California shrimp fishery (see Gillet 2008 for description of gear).

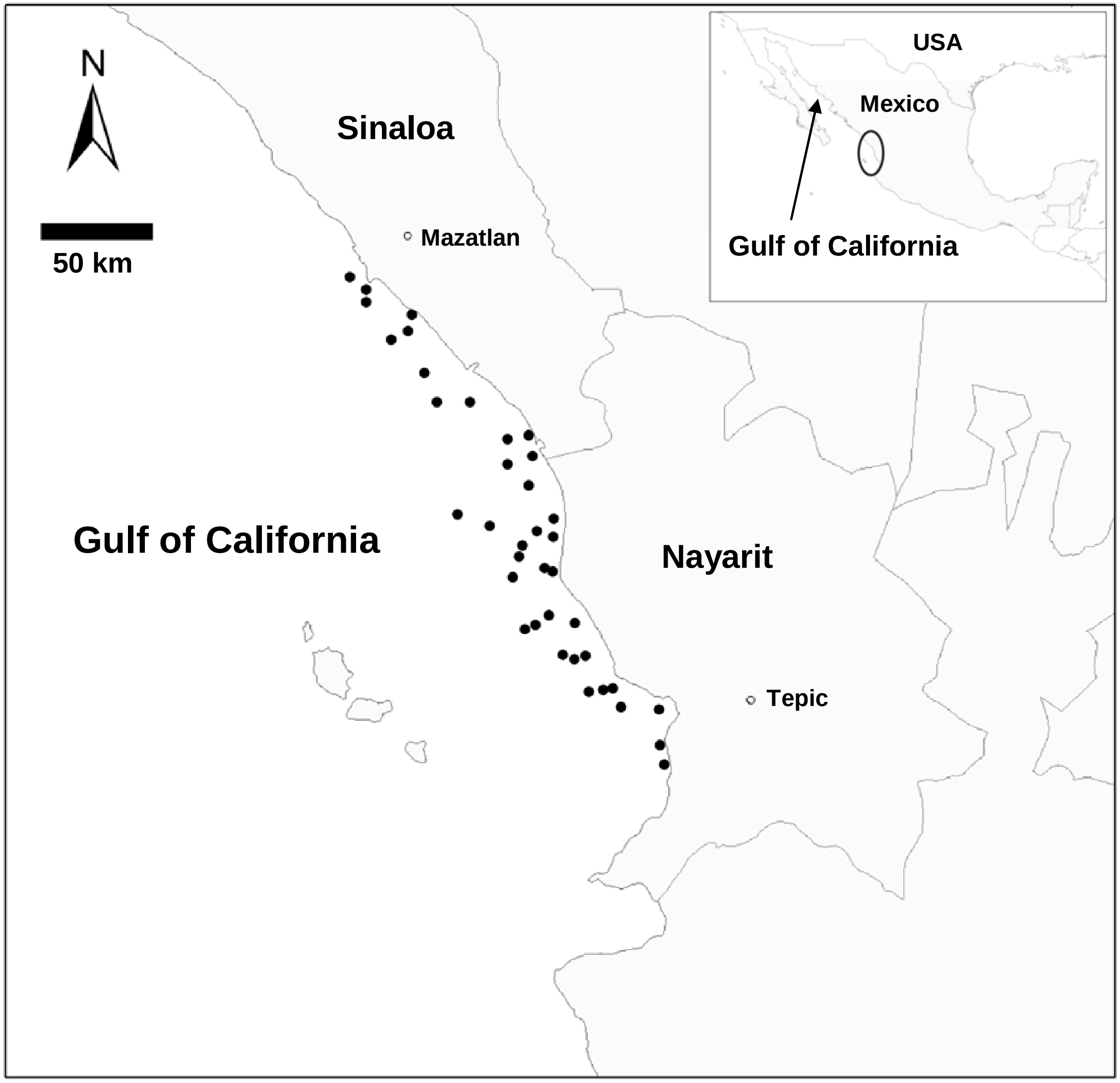

Figure 2.2. Map of Gulf of California (Mexico), indicating sampled area (inset) and station locations.

Tows lasted approximately one hour. Vessel position was tracked using GPS. Depth was held constant throughout the tow. At the end of one hour, fishers landed the catch and estimated its total wet weight (kg). A sample was taken from the catch, before it was sorted by fishers for shrimp and valuable fishes. Average sample wet weight was 26.3 ± 5.3 kg, representing 46 ± 24% of total catch weight. The sample was then sorted for all individuals of the case study taxa, which were frozen in plastic bags for on-land processing.

Samples of each taxon from each station were processed as follows in a fish plant. The total length (L_T, mm) and wet weight (g) of all individuals from each taxon in the sample was measured. Up to 30 individuals from each taxon were haphazardly extracted for further analysis, while trying not to select for size or other characteristics (thereby obtaining a 'representative' sample). Sex and reproductive information were determined through dissection: gonads were removed and sex (male, female), maturity status (immature or mature), and gonad wet weight (g) were recorded. Where it was not possible to determine sex and gonad weight for the smallest juveniles (often), only the maturity status was recorded. Only the ovary was removed for sandperch (see **Reproduction** below). Sample sizes within a given station were insufficient for determining the majority of the life history parameters for the taxa. Thus, for the purposes of this paper, data for each taxon were pooled across stations to increase samples sizes.

Survival and growth

Minimum, maximum and mean total length (L_T) were determined for each of the case study taxa. Mean L_T was compared between sexes using ANOVA to test for evidence of sexual dimorphism.

Length data were pooled across time and space, and binned into 5 mm L_T classes for length-based analyses, including estimation of gear retention, growth parameters, and maturity. Gear retention was defined as the length at which a fish had a 50% chance of being retained by the nets (L_c) (King 2007), and is the size at recruitment to the fishery. To estimate L_c, and the rate at which the curve reaches 100% retention (*r*), a logistic curve was fit to the proportion of individuals caught per length class for each taxon using a non-linear search function in R (R Development Core Team 2004; Crawley 2007). Log likelihood profiling in R was used to determine the 5 and 95% confidence limits for L_c and *r* (Hilborn & Mangel 1997).

For each case study taxon, length frequency data were used to estimate the following growth parameters: number of cohorts; mean L_T of each cohort; proportion of individuals within each cohort; VBGF parameters L_∞ (asymptotic length), and k (rate at which the asymptotic size is approached); and Z (total mortality rate). These parameters were estimated with either parametric mixture analysis (MIX, Pitcher 2002), or non-parametric methods (ELEFAN, SCLA and PROJMAT, Kirkwood et al. 2001), or both.

The details for MIX method can be found in Pitcher (2002), but in summary the method works by fitting normal distributions to grouped data by the method of maximum likelihood. Variable numbers of cohorts were fit to the length frequency data using the SOLVER routine in Excel. Model selection was based on sum of squares (SSQ), which were compared to select the best fit and so the most likely number of cohorts. The introduction of additional parameters, resulting from the addition of cohorts, was penalised by replacing the SSQ by $SSQ(m)(n\text{-}2m)^{-1}$, where n is number of data points, m is number of parameters and $SSQ(m)$ is SSQ for a model with m parameters (Hilborn & Mangel 1997).

The computer program LFDA was used to apply non-parametric length based methods to the length frequency data at 30 day time intervals (Kirkwood et al. 2001). This approach did not, however, prove profitable with the data, as explained below.

Reproduction

Maturity was assessed by both macro and microscopic examination of the gonads. For all taxa, a mature female was deemed an individual whose ovaries contained developed eggs (clearly discernible and opaque), or ready to spawn eggs (completely translucent), or a combination thereof. Since it was not known if the case study taxa were determinate or indeterminate, complete or partial, spawners, individuals were considered mature as long as the ovaries contained developed eggs even if immature or developing eggs were also present. Mature female ovaries occupied ⅔ to full length of body cavity, were orange in colour, and granular in appearance. A mature male had cream to white coloured testes, which were enlarged as opposed to flat and ribbon-like. Milt flowed freely when testes were pinched or pressed.

Maturity status of sandperch was assessed by the ovaries, and not the testes. The female sex organ was much more prominent, and often completely obscured the testes. It therefore proved difficult to separate the two organs without damaging the testes. If the ovary matures after the testes (*cf* Bortone 1971), then size at maturity for this taxon (based on the ovary) would be inflated.

Sex ratio was calculated as the proportion of individuals that were males. Only mature individuals were included in this calculation as juveniles were more likely to be identified as female than male, such that including immature individuals would bias the results towards a higher number of females. A Chi Squared test was used to test if the sex ratio was significantly different from unity.

Length at maturity (L_m) was calculated as the L_T at which 50% of individuals in the population were mature. To estimate L_m, and the rate at which the curve reaches 100% maturity (γ), a logistic curve was fit to the proportion caught per length class for each taxon using a non-linear search function in R (R Development Core Team 2004; Crawley 2007). Likelihood profiling in R was used to determine the 5 and 95% confidence limits for L_m and the slope of the maturity curve (γ) (Hilborn & Mangel 1997).

Gonado-somatic index (I_G) of individual fish, the wet weight of the gonads divided by the wet weight of the intact individual, was plotted against L_T and sample day (where sample day 0 = 25 September 2006), seeking reproductive patterns. The former analysis tests whether fish are investing more in reproduction as they grow, while the latter analysis seeks peaks in reproductive activity. Regression analysis was used to look for relationships between I_G and both L_T and sample day. I_G values were square root (sqrt) transformed to reduce heterogeneity of variance, and satisfy the assumptions of linear regression.

Results

Survival and growth

Size

Estimates of minimum, maximum and mean L_T for each case study taxon were obtained from this study (Table 2.1). Bigscale goatfish, lumptail searobin, and silver stardrum populations were all sexually dimorphic in size. On average, female lumptial searobin and silver stardrum were larger than males (ANOVA, DF = 474, F = 36.67, P < 0.0001; DF = 587, F = 41.97, P < 0.0001, respectively), while males of bigscale goatfish were larger than females (ANOVA, DF = 824, F = 19.77, P < 0.0001).

Previously cited L_{max} values for highfin and inshore sandperch are comparable to the maximum L_T value for sandperch obtained in this study (Table 2.1). The present maximum L_T value for bigscale goatfish is at the lower end of the range of previously cited values of L_{max} for this species (Table 2.1). The present maximum L_T for lumptial searobin was much smaller than other reported values, whereas the maximum L_T value of for silver stardrum obtained in this study was larger than the previously reported L_{max} value (Table 2.1).

Table 2.1. Total length (L_T, mm) of several fish taxa sampled from the catch of two industrial shrimp trawlers in the southern Gulf of California, Mexico, from September 2006 to March 2007. Maximum lengths (L_{max}) from other studies are given for comparison. na = not applicable, SD = standard deviation, N = sample size.

Species	Min L_T	Max L_T	Mean L_T			SD			N			L_{max} (other studies)
	Overall	Overall	Overall	Males	Females	Overall	Males	Females	Overall	Males	Females	
sandperch	55	275	140.64	na	na	28.26	na	na	2037	na	na	highfin sandperch: 260 (Heemstra 1995) inshore sandperch: 237 (Chávez & Arvizu 1972); 280 (Heemstra 1995)
bigscale goatfish	45	219	119.05	152.27	145.46	24.43	20.67	16.92	6803	458	617	213 (Ramos-Santiago et al. 2006); 236 (Lucano-Ramírez et al. 2006); 300 (Eschmeyer et al. 1983)
lumptail searobin	71	185	121.62	120.65	129.39	14.85	13.31	16.72	1110	273	241	280, 320 (as cited in (Mendoza López 2000); 430 (Franke & Acero 1996)
silver stardrum	37	305	113.16	143.67	152.75	30.69	24.28	26.48	5644	413	457	250 (Chao 1995)

Gear retention

Estimates of length at 50% gear retention (L_c, mm) were obtained for all taxa (Figure 2.3). Estimated values of L_c and the rate at which the logistic curve reaches 100% retention (*r*), with corresponding 95% confidence limits and test statistics, were: sandperch (DF = 40) – L_c 124.53 (121.78 – 127.27; t = 93.71), *r* 0.06 (0.05 – 0.07; t = 13.77); bigscale goatfish (DF = 32) – L_c 90.98 (90.65 – 91.31; t = 566.34), *r* 0.23 (0.21 – 0.24; t = 30.96); lumptail searobin (DF = 21) – L_c 104.73 (104.08 – 105.37; t = 340.67), *r* 0.22 (0.20 – 0.25; t = 16.76); silver stardrum (DF = 35) – L_c 82.55 (81.60 – 83.50; t = 179.10), *r* 0.13 (0.12 – 0.14; t = 19.23). All relationships were highly significant ($P < 0.0001$).

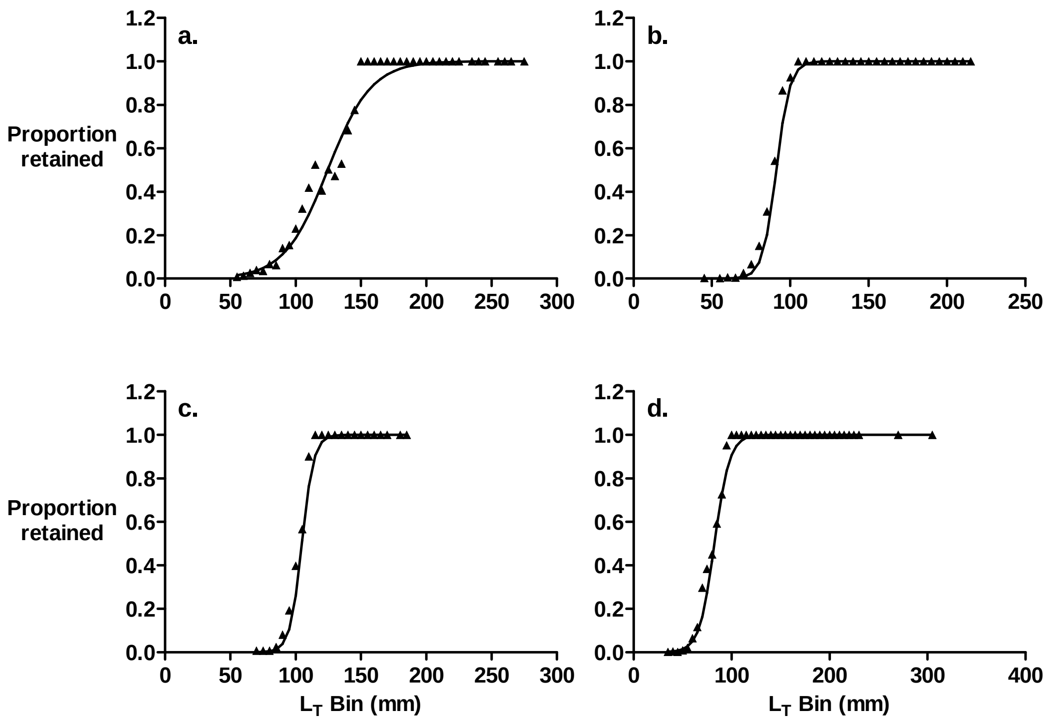

Figure 2.3. Logistic gear retention curves for a) sandperch, b) bigscale goatfish, c) lumptail searobin, and d) silver stardrum, sampled from the catch of two southern Gulf of California (Mexico) industrial shrimp trawlers. Points are observed data, solid line is fitted distribution. Samples were collected from 25 September 2006 to 25 March 2007.

Cohort analysis

Length-frequency histograms suggested a high degree of size overlap among age classes for all taxa (Figure 2.4). Preliminary visual inspection suggested presence of more than one cohort in only two taxa: sandperch, with potential peaks at 115 and 150 mm (Figure 2.4a); and silver stardrum, with potential peaks at 100 and 160 mm (Figure 2.4d).

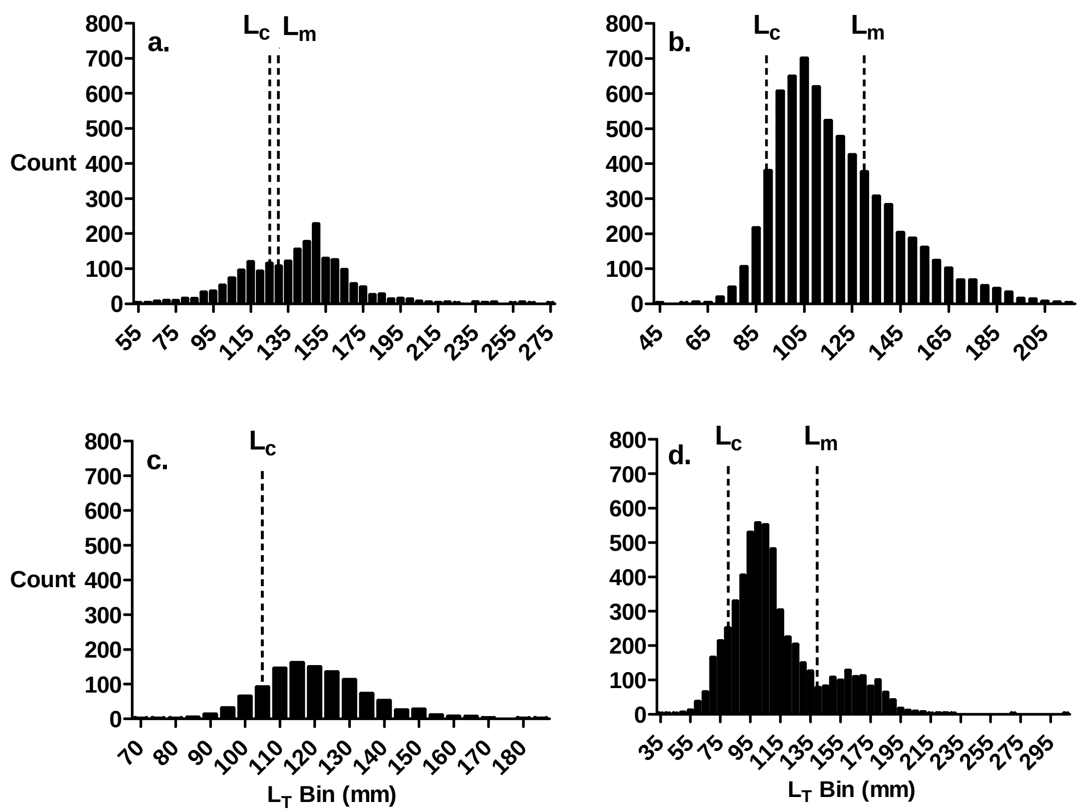

Figure 2.4. Total length (L_T, mm) frequency histograms for a) sandperch, b) bigscale goatfish, c) lumptail searobin, and d) silver stardrum, sampled from the southern Gulf of California (Mexico) shrimp trawl fishery. Bins containing length at 50% retention (L_c) and length at 50% maturity (L_m) are indicated. L_m was not estimated for lumptail searobin as nearly all captured individuals were mature. Samples were collected from 25 September 2006 to 25 March 2007.

The MIX method estimated the number of cohorts that best fit the length frequency data for each taxon (Figure 2.5). MIX identified multiple cohorts in all taxa save lumptail searobin.

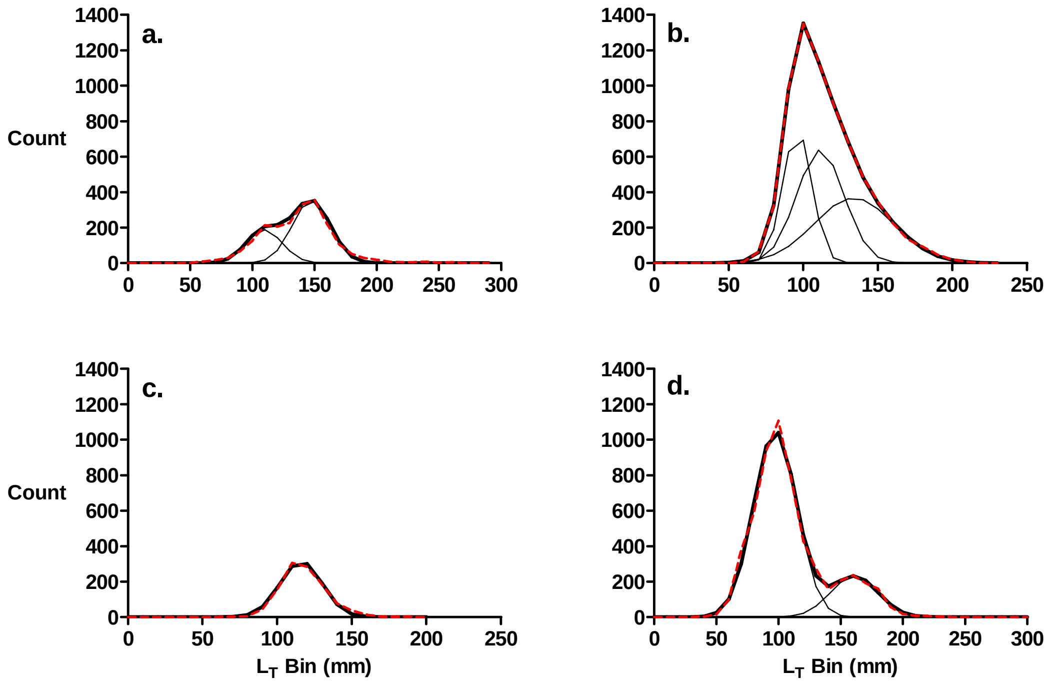

Figure 2.5. Results of length-based analysis (MIX) indicating cohorts fit to length frequency data for samples of a) sandperch, b) bigscale goatfish, c) lumptail searobin, and d) silver stardrum, sampled from the southern Gulf of California (Mexico) shrimp trawl fishery. Solid line indicates fit cohorts. Dashed line is observed data.

For sandperch, two cohorts gave the best fit to the data (Figure 2.5a). The younger cohort had a mean L_T (± standard deviation) of 104.41 ± 14.43 mm, and the older cohort mean L_T 142.39 ± 15.30 mm. The younger cohort contributed fewer individuals to the population than the older (34% and 66%, respectively).

For bigscale goatfish, the best fit model had three cohorts (Figure 2.5b). The youngest cohort had a mean L_T of 90.90 ± 9.51 mm, the middle cohort mean L_T 106.36 ± 15.86 mm, and the oldest cohort mean L_T 128.79 ± 26.63 mm. For this species, the middle and oldest cohorts contributed more individuals to the sample population than the youngest cohort (26, 37 and 36% for youngest, middle and oldest cohorts, respectively). In addition, the standard deviation for the oldest cohort was so wide that it covered the entire length frequency distribution (Figure 2.4b).

The solution for lumptail searobin was only stable when fitting one cohort, even though such a fit did not result in the lowest SSQ (Figure 2.5c). The cohort had a mean L_T of 110.93 ± 14.11 mm.

For silver stardrum the best fit was two cohorts, with the majority of individuals found in the youngest (81% versus 19% for the older cohort) (Figure 2.5d). The younger cohort had a mean L_T of 92.26 ± 17.24 mm, and the older cohort mean L_T 155.90 ± 18.91 mm.

It was not possible to conduct non-parametric length frequency analyses on the data over time (months). ELEFAN, SCLA, and PROJMAT each resulted in poorly defined parameters, unstable solutions (many combinations of L_∞, k and t_0 gave "best fit"), and significant inconsistency among methods. Therefore, results are not presented.

Reproduction

Examination of female gonads indicated that sandperch, bigscale goatfish, and silver stardrum were indeterminate spawners, such that individual ovaries contained eggs at different stages. Lumptail searobin, on the other hand, appeared to be a determinate spawner with all eggs at the same stage within an ovary.

Sex ratios

The ratio of males to females in the sample populations of bigscale goatfish, lumptail searobin and silver stardrum, were not significantly different from unity, indicating equal sex ratios for each of the gonochoristic species (Table 2.2).

Table 2.2. Sampled sex ratios (males/females) for three fish species sampled from the catch of two southern Gulf of California (Mexico) industrial shrimp trawlers. Probability levels (P) are for Chi-squared test.

Species	# Males	# Females	ratio	P
bigscale goatfish	396	430	0.92	0.24
lumptail searobin	251	226	1.11	0.25
silver stardrum	314	295	1.06	0.44

Size at maturity

Estimates of L_m (mm) were obtained for sandperch, bigscale goatfish and silver stardrum (Figure 2.6). Estimated values of L_m and γ, with corresponding 95% confidence limits and test statistics, were: sandperch (DF = 39) – L_m 131.43 (127.57 – 133.32; t = 143.00), γ 0.13 (0.10 – 0.16; t = 9.76); bigscale goatfish (DF = 29) – L_m 135.20 (133.99 – 136.34; t = 237.42), γ 0.13 (0.11-0.15; t = 15.62); silver stardrum (DF = 34) – L_m 137.30 (135.28 – 139.33; t = 141.46), γ 0.08 (0.07 – 0.10; t = 13.99). All regressions were highly significant ($P < 0.0001$). Because the majority (75%) of lumptail searobin in the samples were mature (Figure 2.6c), it was not possible to model L_m for this species. The minimum L_T of a mature individual for this species was 90 mm.

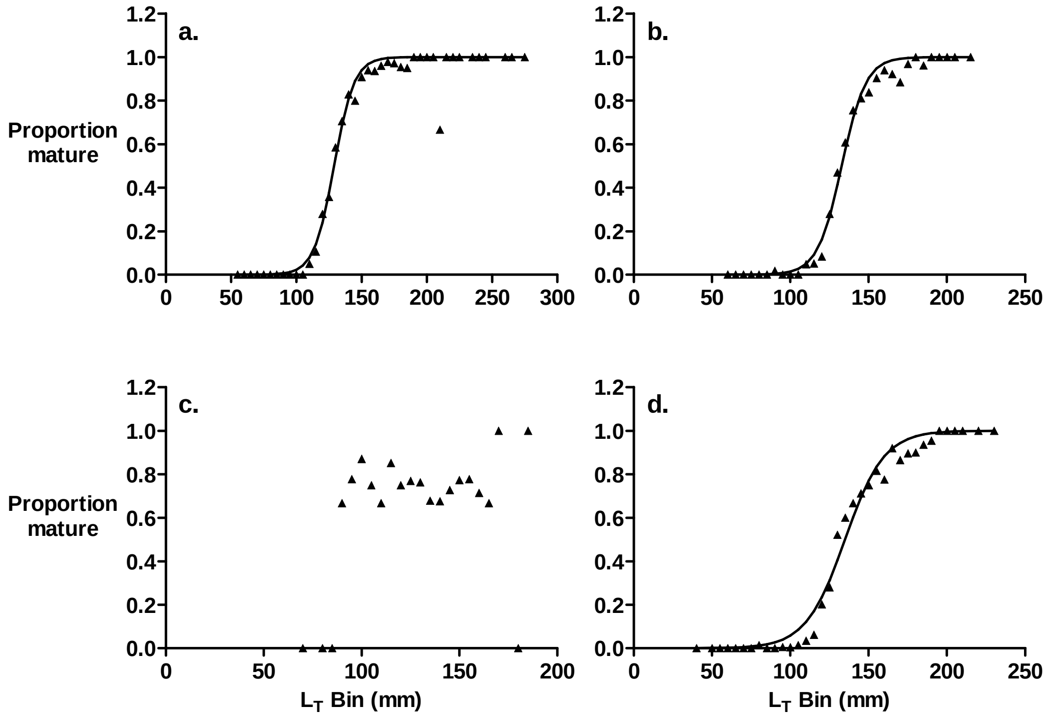

Figure 2.6. Logistic maturity curves for a) sandperch, b) bigscale goatfish, c) lumptail searobin, and d) silver stardrum, sampled from the catch of two southern Gulf of California (Mexico) industrial shrimp trawlers. Points are observed data, solid line is fitted distribution. Samples were collected from 25 September 2006 to 25 March 2007.

Size specific gonado-somatic index

Regression analyses were conducted to determine if and how I_G was related to L_T for each of the case study taxa (Table 2.3). A regression of I_G on L_T was significant for all except males of silver stardrum. For each significant relationship, however, L_T explained less than 25% of the variation in I_G, and for all taxa there were a very wide range of possible I_G values for a given body size. Analysis for lumptail searobin was conducted on mature individuals only, as very few immature individuals were captured. In general, I_G increased with increasing L_T except for lumptail searobin, where I_G decreased with increasing L_T for both males and females. The relationship between I_G and L_T was highly variable for the case study taxa, evidenced by the coefficients of variation (CV) in Table 2.3 (distributions with CV > 1 are considered high-variance, Frank 1995).

Table 2.3. Transformed (square root) gonado-somatic index (I_G) of a) testes and b) ovaries, regressed against total length (L_T, mm) for several fish species sampled from the catch of two southern Gulf of California (Mexico) industrial shrimp trawlers. DF = degrees of freedom, β = regression coefficient, int = intercept, SE = standard error, CV = coefficient of variation, R^2 = coefficient of determination, probability levels (P) are for F-test.

Species	DF	β	SE	int	SE	CV	R^2	F	P
a.									
bigscale goatfish	381	1.61E-04	2.53E-05	0.022	0.004	2.47	0.10	40.52	<0.0001
lumptail searobin	246	-3.25E-04	1.04E-04	0.143	0.013	-5.02	0.04	9.76	0.002
silver stardrum	267	8.14E-07	5.17E-05	0.047	0.008	1038.52	0.00	2.48E-04	0.99
b.									
sandperch	1127	2.04E-04	7.23E-05	0.092	0.011	11.91	0.01	7.95	0.005
bigscale goatfish	578	1.26E-03	9.56E-05	-0.053	0.014	1.77	0.23	172.72	<0.0001
lumptail searobin	222	-1.07E-03	2.19E-04	0.402	0.028	-3.04	0.10	23.95	<0.0001
silver stardrum	407	7.00E-04	8.75E-05	0.012	0.014	2.53	0.13	59.58	<0.0001

Reproductive cycles

Mature individuals of all taxa were found throughout the sampling period (September to March) (Figure 2.7). The proportion of mature sandperch and silver stardrum was variable, the proportion of mature bigscale goatfish decreased, and the proportion of mature lumptail searobin increased, over the course of sampling (Figure 2.7).

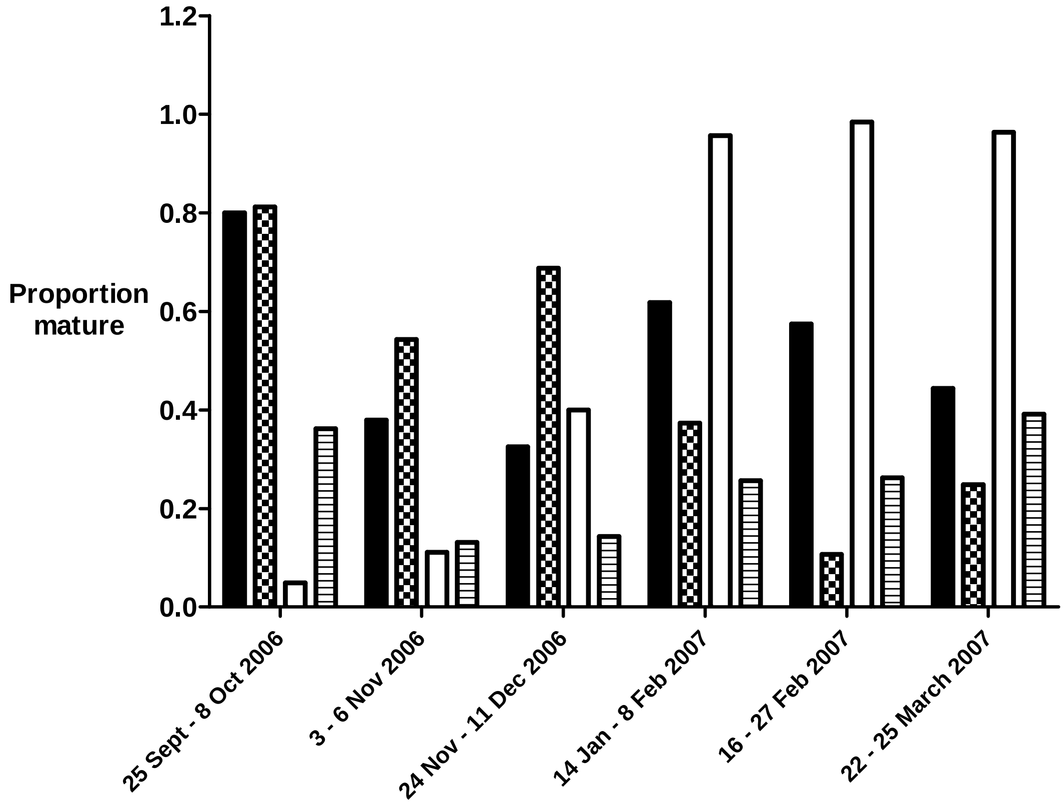

Figure 2.7. The proportion of mature individuals for several fish species, in a given 30 day period, captured during six months of sampling onboard two southern Gulf of California (Mexico) industrial shrimp trawlers (start date: 25 September 2006; end date: 25 March 2007). Solid = sandperch, checkered = bigscale goatfish, clear = lumptail searobin; striped = silver stardrum.

A regression of I_G against sample day was significant for all except females of bigscale goatfish (Table 2.4). Similar to L_T, however, sample day explained less than 25% of the variation in I_G for all groups, save male silver stardrum, where sample day explained 40% of the variation in I_G. For female silver stardrum, the relationship between I_G and sample day was improved when only mature females were included in the analysis. In this case, sample day explained 36% of variation in I_G (Table 2.4b). The relationship between I_G and sample day was highly variable for the case study taxa, evidenced by the coefficients of variation (CV) in Table 2.4, which are all greater than one.

Table 2.4. Transformed (square root) gonado-somatic index (I_G) of a) testes and b) ovaries, regressed against sample day (where 0 = 25 September 2006) for several fish species sampled from two southern Gulf of California (Mexico) industrial shrimp trawlers. DF = degrees of freedom, β = regression coefficient, int = intercept, SE = standard error, CV = coefficient of variation, R^2 = coefficient of determination, probability levels (P) are for F-test.

Species	DF	β	SE	int	SE	CV	R^2	F	P
a.									
bigscale goatfish	381	5.55E-05	8.86E-06	0.041	0.001	3.12	0.09	39.25	<0.0001
lumptail searobin	264	2.97E-04	4.08E-05	0.059	0.006	2.23	0.17	53.11	<0.0001
silver stardrum	267	1.52E-04	1.15E-05	0.033	0.001	3.40	0.40	174.82	<0.0001
b.									
sandperch	1127	1.70E-04	3.38E-05	0.101	0.005	6.69	0.02	25.19	<0.0001
bigscale goatfish	578	1.46E-05	3.03E-05	0.129	0.004	49.93	0.00	0.23	0.63
lumptail searobin	237	9.59E-04	1.11E-04	0.120	0.016	1.78	0.24	74.83	<0.0001
silver stardrum	415	2.28E-04	3.17E-05	0.093	0.004	2.43	0.11	51.73	<0.0001
silver stardrum[1]	272	3.35E-04	2.72E-05	0.105	0.003	0.57	0.36	151.78	<0.0001

[1]only mature individuals were included in the analysis

Discussion

Fisheries dependent sampling provided insight into the life history and population parameters of several small fish species obtained as bycatch in the southern Gulf of California shrimp trawl fishery, and allowed for preliminary assessment of potential for impact resulting from their capture. The present study raises concern for the sampled populations of bigscale goatfish and silver stardrum. The majority of both these species were caught before getting the chance to mature, although prolonged periods of reproduction, which may peak during the fishery's closed season, may offer some protection. The majority of sandperch were caught after length at maturity, and also demonstrated a prolonged period of reproduction. This would seem to suggest a lower potential for impact by the fishery. However, hermaphroditism has been suggested to increase vulnerability (Stobutzki et al. 2001). Finally, the observed patterns for lumptail searobin suggest the potential for adverse fishing effects in that the majority of captured individuals were fully mature, and females had not yet spawned – but also potential for buffering these effects as smaller/younger individuals appeared to be located outside the trawl grounds, such that young develop away from the fishing pressure.

Size

In theory, one can compare L_{max} estimates across studies to deduce possible impacts of fishing (Heino & Godø 2002). A smaller than expected maximum size may be an indication of adverse fishing effects (Heino & Godø 2002), or suggest incomplete recruitment of older age classes to the fishery (King 2007). The latter may occur if older fish are outside the sample area, or able to avoid capture. Since the vast majority of captured individuals of lumptail searobin were mature and reproducing, one might infer that the smaller L_{max} of the sampled population in this study compared to others could be a fishing effect. Indeed, a decrease in average size and length at maturity, corresponding to an increase in exploitation rate, has already been observed for lumptail searobin in Peru (Castillo-Rojas et al. 2000). In reality, however, comparisons of L_{max} across studies are fraught with problems. Fish populations often show a distribution of size with depth, and larger/older individuals are less numerous than smaller/younger ones. Therefore, to obtain a sample L_{max} close to true population L_{max}, one would have to sample quite heavily across the entire depth range of the population. The large sample sizes in the present study may, therefore, explain the larger than previously cited L_{max} obtained for silver stardrum.

Very little can be inferred from the sexual dimorphism found in all gonochoristic species in this study. The size differences (an average of 8 mm) did not seem large enough to suggest differing rates of gear retention by sex, particularly given the equal sex ratios in samples for all the gonochoristic species.

Survival and growth

This study of species with small adult sizes encountered many limitations with respect to analyses of survival and growth. For example, the classic assumption that selectivity of trawl nets follows a logistic curve, with greater retention of larger fish within a species (King 2007), could not be made. Thus, the estimates for gear retention can not be used to determine catchability (a required parameter for converting catch to biomass, Hilborn & Walters 1991). One can, however, compare what is being retained to important length based reference points, such as length at maturity (addressed under **Risk assessment** below).

It was also not possible to use parametric length based growth analyses to estimate growth and mortality. Previous research suggests that each of the fitted cohorts in this study contain individuals of multiple ages, and thus do not represent separate age classes. Studies on lumptail searobin have cited maximum age at 9 years (Schmitter-Soto & Castro-Aguirre 1991), on inshore sandperch have cited maximum age at 12 years (Chávez & Arvizu 1972), and the authors data on silver stardrum suggest a maximum age of at least 8 years (Appendix I). Since the fitted cohorts could not be assumed to represent age classes – perhaps because of continuous reproduction in three taxa – the results could not be used to estimate VBGF parameters, or mortality rates (King 2007). Continuous or prolonged periods of recruitment lead to large variance in length by age, and therefore obscure cohorts from young ages on (Pitcher 2002). Understanding growth, and then mortality, for these taxa will require analysis of hard parts (e.g. otoliths and scales).

Sampling for this study may be non-representative with respect to population structure. A true population structure should contain the greatest proportion of individuals in the first cohort, and diminishing numbers of individuals in subsequent cohorts, yet length-based analysis for sandperch and bigscale goatfish found fewer individuals in the first cohort than subsequent cohorts. This may provide evidence of non-representative sampling (such that the first age classes are missing), but may also result from a high degree of overlap in size among age group due to continuous reproduction. The samples contained almost no immature lumptail searobin. Therefore, the single cohort fitted to the length data for this species probably represents the pile up effect of older ages with the younger/smaller fish lying elsewhere.

Reproduction

Comparing temporal patterns in reproduction and fishing pressure may indicate the potential for adverse fishing effects (Andrew & Pepperell 1992). Reliable L_m estimates were obtained for three of the case study taxa: sandperch, bigscale goatfish and silver stardrum. If the ovary matures after the testes in sandperch (*cf* Bortone 1971), then L_m for this taxon (based on the ovary) would be overestimated. The estimate for bigscale goatfish was smaller than in the literature (183 mm, Lucano-Ramírez et al. 2006), but similar to that derived from empirical relationships with L_{max} (141 mm; $\log L_m = 0.84(\log L_{max}) - 0.06$; $R^2 = 0.81$, Froese & Binohlan 2000). Size at maturity could not be modelled for lumptail searobin, as the majority of sampled individuals were mature. But L_{max} of the sampled lumptail searobin translates into an L_m of 92 mm (using equation from Froese and Binohlan 2000), similar to the size of the smallest mature individual in the samples, 90 mm. Size at maturity was estimated to be much larger for a Peruvian population of lumptail searobin (202 mm, Samamé & Fernández 2000). All populations appeared to reach maturity quite quickly, as is common in many fish species (Chen & Paloheimo 1994).

Mature individuals of sandperch, bigscale goatfish and silver stardrum were found in all months studied, and ovaries contained eggs at differing stages of development, suggesting extended periods of reproduction. In contrast, eggs within a given ovary of lumptail searobin were all at the same stage, leaving open the question of whether they are total spawners or batch spawners. All lumptail searobin caught early in the season (September to December) contained immature eggs, whereas all captured individuals during the second half of the season (January to March) contained mature eggs, even though there was complete size overlap. It may be that individuals produced one batch of eggs per breeding season, with eggs fully mature by January, and that truly immature individuals (i.e. those incapable of producing mature eggs) are found outside the trawl grounds. In that case, the trawl fishery might pose problems by removing individuals before they can spawn, but at the same time leave the young alone. The consequences would depend on whether juvenile survival or fecundity is the limiting factor in population viability for this species (Walters & Martell 2004).

With only half a year of samples, it was not possible identify reproductive peaks for these populations. It does appear, however, that reproductive activity was increasing through the winter into spring (September to March), and might peak sometime in the spring/summer months (April – August), just when the Gulf's industrial trawl fishery is closed. Elsewhere, reproductive peaks for bigscale goatfish in southern Mexico occurred in summer (August – October, Ramos-Santiago et al. 2006), and Peruvian lumptail searobin showed peak periods of reproduction in spring and summer (November to April, Castillo-Rojas et al. 2000; Samamé & Fernández 2000). Readers should note that the coefficients of determination (R^2) were very low for the majority of regressions between I_G and both L_T and sample day, probably because highly variable I_G values, consistent with batch spawning, obscured stronger relationships. Although low correlation values limit the explanatory power of these relationships, the results are still useful in providing preliminary insight into potential for impact from the trawl fishery on these taxa, and reflect relationships worthy of further investigation.

Example risk assessment

Sufficient information on size and maturity was obtained on the case study taxa to assess their incidental capture against Froese's length-based indicators of overfishing (2004). The first, "let them spawn", is the percent of retained individuals that are less than size at maturity. The second, "let them grow", is the percent of retained individuals that are outside ± 10% of optimum length (L_{opt}), where L_{opt} is the length at which the maximum yield can be obtained. It should be noted that this indicator is only directly relevant to incidentally captured fishes of commercial value. L_{opt} was calculated from L_m [$\log L_{opt} = 1.05(\log L_m - 0.06$, $R^2 = 0.89$, Froese & Binohlan 2000]. The third, "let the mega-spawners live", is the percent of fish in the catch that are greater than L_{opt} + 10%. A justification for this last indicator is that larger females are disproportionately more fecund. Since a significant positive relationship was observed between I_G and L_T in females of sandperch, bigscale goatfish and silver stardrum, this indicator may be appropriate for these species. However, I_G may be a poor proxy for egg number in partial spawners. Ideally (though perhaps unrealistically), each of these indicators would be near 0%.

Based on the three categories posed by Froese, the present findings suggest room for management concern. The values for sandperch are not problematic, with about a third (37%) of captured individuals at less than L_m, and about half (55%) lying outside the range of L_{opt} values for this species. A further 16% of the capture individuals were in the “mega-spawner” size range. In contrast, the indicator values for bigscale goatfish and silver stardrum do suggest potential for overfishing of these populations. Approximately 80% of retained individuals of these two species were smaller than L_m (79 and 81%, respectively). This suggests that the majority of individuals are not getting the opportunity to reproduce before capture, and therefore the potential for overfishing (Hilborn & Walters 1991). The majority of individuals were also outside the range of L_{opt} values for the species (84 and 89%, respectively). Only a small number of retained individuals were considered “mega-spawners” (4 and 8%, respectively). The small proportion of larger/older fish in the samples could be due to larger individuals of these species being outside the sample area, but also because high adult mortality rates (partly due to fishing) means there are few larger/older individuals in the population. Erosion of a populations age-structure reduces its resilience (Froese 2004). Finally, the majority of retained individuals of lumptail searobin were outside the range of L_{opt} values (82%), and the majority of these were considered mega-spawners (81%). However, as most of the captured individuals were mature (75%), this may offset the effects of removal of the potentially more fecund females. Further research is needed on the reproductive patterns of this species.

General comments

The results of this fisheries dependent study suggest that correlative approaches for assessment based on knowledge of life history and population parameters can help provide a pragmatic option for preliminary evaluation of the impacts of shrimp trawls on the small fish species they catch incidentally. This implies that appropriate life history information, where it can be deduced, will provide insight into trawling impacts through length based and qualitative approaches. In addition, the study hints that that at least some small fishes show potential for overfishing, calling for a re-evaluation of the status of small species, which have generally been considered resilient to fishing pressures (e.g. Jennings et al. 1998; Jennings et al. 1999a; Jennings et al. 1999b).

While the analytical approaches employed here are useful for suggesting species of potential concern, fisheries independent sampling and/or long-term data would be needed to confirm impact. The fact that bigscale goatfish and silver stardrum are two of the most commonly caught small fishes in this fishery might lead to the inference they have actually been resilient to fishing pressure, contrary to the indications of this study. Indeed, it is certainly true that impacts from trawl fisheries on small fishes need not be negative. For example, decreases in predators might allow small fish populations to increase, and trawling induced changes to benthic communities and/or food subsidies from discarding practices might variably affect the diet of small fishes (Jennings & Kaiser 1998; Fonds & Groenewold 2000; Greenstreet & Rogers 2000; Hiddink et al. 2008). Unfortunately, without a time series of data one cannot deduce the population trends for the case study fishes. Even if bigscale goatfish and silver stardrum are now abundant relative to other species in the catch, their populations may still be much smaller than in the past.

A final thought for consideration. In Mexico, the small fish component of the shrimp trawl bycatch is considered as one group for management purposes. Fishers need only estimate the total weight of the bycatch for their log books – collectively termed *fauna* or *basura* (translated: fauna or trash). Comparing among the case study taxa indicates that small fish species obtained as bycatch in the Gulf of California shrimp trawl fishery have different life history and population characteristics, and so probably differ in their vulnerabilities to fishing pressure. Thus, monitoring and managing them as a group may mask species-specific issues. The combined catch of small fish could remain stable (or even increase), while the species composition is changing.

Literature cited

Alverson, D. L., M. H. Freeberg, J. G. Pope, and S. A. Murawski. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper No. 339: 1-233.

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

Arvizu-Martinez, J. 1987. Fisheries activities in the Gulf of California, Mexico. CalCOFI Report **XXVIII**: 32-36.

Bergman, M. J. N., and J. W. van Santbrink. 2000. Fishing mortality of populations of megafauna in sandy sediments. Pages 49-68 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Bortone, S. A. 1971. Studies on the biology of the sand perch, *Diplectrum formosum* (Perciformes: Serranidae). State of Florida Department of Natural Resources Technical Series **65**: 1-27.

Braccini, J. M., B. M. Gillanders, and T. I. Walker. 2006. Hierarchical approach to the assessment of fishing effects on non-target condricthyans: case study of *Squalus megalops* in southeastern Australia. Canadian Journal of Fisheries and Aquatic Sciences **62**: 2456-2466.

Castillo-Rojas, R., E. Gómez, and F. Paredes. 2000. Algunos aspectos adicionales sobre la biologia y pesqueria del "falso volador", *Prionotus stephanophrys*. Informe Instituto del Mar del Peru **127**: xx-xx.

Chao, L. N. 1995. Sciaenidae. Corvinas, barbiches, bombaches, corvinatas, corvinetas, corvinillas, lambes, pescadillas, roncachos, verrugatos. Pages 1427-1518 in W. Fischer, F. Krupp, W. Schneider, C. Sommer, K. E. Carpenter, and V. Niem, editors. Guía FAO para Identificación de Especies para lo Fines de la Pesca. Pacifico Centro-Oriental. FAO, Rome.

Chávez, H., and J. Arvizu. 1972. Estudio de los recursos pesqueros demersales del Golfo de California, 1968-1969: III. Fauna de acompañamiento del camarón (peces finso y "basura"). Pages 361-378 in J. Carranza, editor. Memorias del IV Congreso Nacional de Oceanografía, México.

Chen, Y., and J. E. Paloheimo. 1994. Estimating fish length and age at 50% maturity using a logistic type model. Aquatic Sciences **56**: 206-219.

Crawley, M. J. 2007. The R Book. John Wiley & Sons Ltd., Chichester, UK.

Diamond, S. L., L. B. Cowell, and L. B. Crowder. 2000. Population effects of shrimp trawl bycatch on Atlantic croaker. Canadian Journal of Fisheries and Aquatic Sciences **57**: 2010-2021.

Diamond, S. L., L. B. Crowder, and L. B. Cowell. 1999. Catch and bycatch: The qualitative effects of fisheries on population vital rates of Atlantic croaker. Transactions of the American Fisheries Society **128**: 1085-1105.

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

Enríquez-Andrade, R., G. Anaya-Reyna, J. C. Barrera-Guevara, M. d. l. Á. Carvajal-Moreno, M. E. Martínez-Delgado, J. Vaca-Rodríguez, and C. Valdés-Casillas. 2005. An analysis of critical areas for biodiversity conservation in the Gulf of California region. Ocean and Coastal Management **48**: 31-50.

Eschmeyer, W. N., E. S. Herald, and H. Hammann 1983. A field guide to Pacific coast fishes of North America. Houghton Mifflin Company, Boston.

Fonds, M., and S. Groenewold. 2000. Food subsidies generated by the beam-trawl fishery in the southern North Sea. Pages 130-150 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Frank, H. a. A., S.C. 1995. The coefficient of variation. Pages 58-59. Statistics: Concepts and Applications. Cambridge University Press, Cambridge.

Franke, R., and A. Acero. 1996. Peces oseos comerciales del Parque Gorgona, Pacifico colombiano (Osteichthyes: Muraenesocidae, Hemiramphidae, Belonidae, Scorpaenidae, Triglidae, Malacanthidae, Gerreidae, Sparidae, Kyphosidae, Sphyraenidae e Istiophoridae). Revista de Biologia Tropical **44**: 763-770.

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

Froese, R., and C. Binohlan. 2000. Empirical relationships to estimate asymptotic length, length at first maturity and length at maximum yield per recruit in fishes, with a simple method to evaluate length frequency data. Journal of Fish Biology **56**: 758-773.

García-Caudillo, J. M., and J. V. Gómez-Palafóx 2005. La pesca industrial de camarón en el Golfo de California: Situación económico-financiera e impactos socio-ambientales. Conservación Internacional-Región Golfo de California, Guaymas.

Garcia, S. M., and R. J. R. Grainger. 2005. Gloom and doom? The future of marine capture fisheries. Philosophical Transactions of the Royal Society of London B **360**: 21-46.

Greenstreet, S. P. R., and S. I. Rogers. 2000. Effects of fishing on non-target fish species. Pages 217-234 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Heemstra, P. C. 1995. Serranidae. Meros, serranos, guasetas, enjambres, baquetas, indios, loros, gallinas, cabrillas, garropas. Pages 1565-1613 in W. Fischer, F. Krupp, W. Schneider, C.

Sommer, K. E. Carpenter, and V. Niem, editors. Guía FAO para Identificación de Especies para lo Fines de la Pesca. Pacifico Centro-Oriental. FAO, Rome.

Heino, M., and O. R. Godø. 2002. Fisheries-induced selection pressures in context of sustainable fisheries. Bulletin of Marine Science **70**: 639-656.

Hiddink, J. G., A. D. Rijnsdorp, and G. Piet. 2008. Can bottom trawling disturbance increase food production for a commercial fish species? Canadian Journal of Fisheries and Aquatic Sciences **65**: 1393-1401.

Hilborn, R., and M. Mangel 1997. The Ecological Detective: Confronting Models with Data. Princeton University Press, Princeton, NJ.

Hilborn, R., and C. J. Walters 1991. Quantitative Fisheries Stock Assessment: Choices, Dynamics and Uncertainties. Chapman and Hall, New York.

Jennings, S., and M. J. Kaiser. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology **34**: 201-352.

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

Kelleher, K. 2004. Discards in the world's marine fisheries: An update. FAO, Rome.

King, M. 2007. Fisheries Biology, Assessment and Management. Blackwell Publishing, Oxford, UK.

Kirkwood, G. P., R. Auckland, and S. J. Zara. 2001. Length Frequency Distribution Analysis (LFDA), Version 5.0. MRAG Ltd., London, UK.

Lucano-Ramírez, G., S. Ruiz-Ramírez, and J. A. Rojo-Vázquez. 2006. Composición por tallas y ciclo reproductivo de *Pseudupeneus grandisquamis* (Pices: Mullidae) en el Pacifico central Mexicano. Revista de Biologia Tropical **54**: 195-207.

Manjarrez Acosta, C. 2001. Variación estacional de la comunidad de peces de fondos blandos del área costera del sur de Sinaloa. MSc. Instituto Politécnico Nacional, La Paz, BCS, Mexico.

Mendoza López, J. d. D. 2000. Crecimiento de *Prionotus stephanophrys* "Trigla coronada" (Lockington, 1880) en la costa oriental del Golfo de California, México. MSc. Instituto Politécnico Nacional, La Paz, BCS, Mexico.

Naylor, R. L., R. J. Goldburg, J. H. Primavera, N. Kautsky, M. C. M. Beveridge, J. Clay, C. Folke, J. Lubchenco, H. Mooney, and M. Troell. 2000. Effect of aquaculture on world fish supplies. Nature **405**: 1017-1024.

Ortiz, M. 1998. Shrimp bycatch in the US Gulf of Mexico: Review and evaluation of bycatch effects on exploited fish stocks. PhD. University of Miami, Coral Gables, Florida.

Pauly, D., J. Dalsgaard, V. Christensen, R. Froese, and F. J. Torres. 1998. Fishing down marine food webs. Science **279**: 860-863.

Pitcher, T. J. 2002. A bumpy old road: size-based methods in fisheries assessment. Pages 188-210 in P. J. B. Hart, and J. D. Reynolds, editors. Handbook of Fish and Fisheries, Volume 2. Blackwell Publishing, Oxford.

Pope, J. G., D. S. Macdonald, N. Daan, J. D. Reynolds, and S. Jennings. 2000. Gauging the impact of fishing mortality on non-target species. ICES Journal of Marine Science **57**: 689-696.

R Development Core Team. 2004. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ramos-Santiago, E., J. M. Ramirez-Gutierrez, R. Mendoza-Rodriguez, and M. Tapia-Garcia. 2006. Reproducción, distribución y abundancia del pez *Pseudupeneus grandisquamis* (Perciformes: Mullidae), en el Golfo de Tehuantepec, México. Revista de Biologia Tropical **54**: 1103-1112.

Samamé, M., and F. Fernández. 2000. Evaluación biológico pesquera del "falso volador" *Prionotus stephanophrys* Lockington, componente de la ictiofauna demersal del Perú. Informe Instituto del Mar del Peru **153**: xx-xx.

Schmitter-Soto, J. J., and J. L. Castro-Aguirre. 1991. Edad y crecimiento de *Prionotus stephanophrys* (Osteichthyes: Triglidae) en la costa occidental de Baja California Sur, México. Revista de Biologia Tropical **39**: 23-29.

Schmitter-Soto, J. J., and J. L. Castro-Aguirre. 1994. Age and growth of three searobins (Pices: Triglidae) off the western coast of Baja California Sur, México. Revista de Biologia Tropical **42**: 271-279.

Schneider, M. 1995. Mullidae. Salmonetes. Pages 1299-1300 in W. Fischer, F. Krupp, W. Schneider, C. Sommer, K. E. Carpenter, and V. Niem, editors. Guía FAO para Identificación de Especies para lo Fines de la Pesca. Pacifico Centro-Oriental. FAO, Rome.

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

Walters, C. J., and S. J. D. Martell 2004. Fisheries Ecology and Management. Princeton University Press, Princeton, NJ.

3. Divergent distribution patterns mean that diverse trawl closures are needed to address small fish bycatch in tropical shrimp trawl fisheries [2]

[2] A version of this chapter has been submitted for publication. Foster, S.J. Using distribution data to address small fish bycatch in tropical shrimp trawl fisheries.

Introduction

Commitments to ecologically sound fisheries management (e.g. Sherman et al. 2005), and future food security (e.g. Delgado et al. 2003) require that small fish bycatch is held to sustainable levels. Bycatch is the non-target part of the catch, and is either discarded at sea or retained for human or animal consumption (Andrew and Pepperell 1992). Species taken as bycatch may be harvested at unsustainable levels (even in regulated fisheries), threatening species diversity and ecosystem health (Hall 1996). Shrimp trawlers catch many million tonnes of non-target species each year (Alverson et al. 1994; Kelleher 2004), with tropical shrimp fisheries having the highest discard rates (>27% of total discards, Kelleher 2004). The majority of bycatch species in tropical shrimp fisheries are comprised of small fishes that mature at less than 20 cm (Alverson et al. 1994; Kelleher 2004). Despite this disproportionate impact, research into mitigating the effects of trawling has been focused in temperate or developed tropical systems, and on charismatic or commercially important species (S.J.F., unpublished data).

While technological advances have helped reduce bycatch of many groups of fauna (Broadhurst 2000), bycatch of small fish species is still a problem, primarily because they are the same size as tropical shrimps (Clucas 1997; Cochrane 2002). Most bycatch reduction devices exclude individuals that are larger than target shrimps (Broadhurst 2000; Eayrs 2007). Ongoing work continues to explore the use of grids and other devices mounted in the net to allow smaller fish and shrimps to escape (Eayrs 2007). Specifically, Juvenile and Trash Excluder Devices (JTED) have been developed and tested widely throughout South East Asia with good success, but high construction costs may limit implementation (Eayrs 2007) (NB: 'trash fish' is a term applied to small fish species of no commercial value). In spite of technological advances, bycatch levels are still high, with discard rates averaging 56% across tropical shrimp fisheries (Kelleher 2004). It may never be possible to find a technological solution that works for all small fishes, because of their diverse behaviour patterns, physiological conditions, and morphologies (Bublitz 1995; Loverich 1995).

Spatio-temporal closures that reduce trawling intensity are the most likely means to adjust bycatch of small fishes. Managed access or closures can be useful for protecting a variety of fishes with divergent life history strategies, including the hundreds of small fish species caught by tropical shrimp trawlers (Winemiller & Rose 1992). Trawling restrictions reduce incidental fishing mortality, but also habitat damage, thereby addressing both direct and indirect impacts on bycatch species in the closed area (Kaiser 2000; Stefansson & Rosenberg 2005; Kaiser et al. 2006). Including a temporal element to trawl restrictions may ease implementation. Limiting closures to periods of high fish abundance and/or vulnerability might reduce bycatch without significant economic costs for the fishery (Andrew & Pepperell 1992).

Information on a species' distribution, migration and spawning seasons is required to assess the potential value of spatio-temporal restrictions in mitigating fisheries' impacts (Morgan & Chuenpagdee 2003). With these types of information one could, in theory, exclude fishing from areas of critical habitat – such as spawning or feeding grounds, or during periods of high vulnerability – such as reproductive peaks. One could also map spatio-temporal distributions of fishing effort onto species distribution to locate 'bycatch hotspots', regions of greatest overlap, and thus locations where managed areas would have greatest potential for impact.

Unfortunately, few life history data are available for the small fishes obtained as bycatch in tropical shrimp trawls (e.g. Stobutzki et al. 2001). Assessing species distributions can be costly, as collecting the necessary biological data requires time and resource intensive field collections and studies (Stockwell & Peterson 2002). Since the majority of small fishes obtained as bycatch in tropical shrimp trawls have little to no commercial value (especially compared to targeted shrimp) and are not charismatic enough to warrant public attention, there may be no real incentives to conduct these types of surveys. However, some of these costs may be reduced by taking advantage of fisheries dependent sampling opportunities.

This study derives distribution data that help evaluate strategies to reduce small fish bycatch in tropical shrimp trawl fisheries. Fisheries dependent research was conducted for two small fish species obtained as bycatch in the southern Gulf of California (Mexico) shrimp trawl fishery, and the results used to consider how the resulting data can help inform management. Of particular focus is assessing potential for impacts from trawling, and the potential use of spatio-temporal trawl restrictions to mitigate impact for these species.

Methods

Study site

This work focused on the southern Gulf of California, one the Gulf's three biogeographic regions (Walker 1960; Figure 3.1). More than 1000 large industrial shrimp trawlers sweep an area of the sea floor equivalent to two times that of the Gulf on an annual basis (Brusca et al. 2005). A decrease in diversity and biomass of bycatch species over the last 50 years has been inferred from limited scientific and anecdotal information (Brusca et al. 2005; Sáenz-Arroyo et al. 2005; Sáenz-Arroyo et al. 2006; Lozano-Montes et al. 2008).

Study species

Two commonly caught small fishes in the Gulf's trawl fishery were chosen as case studies for this research, bigscale goatfish (*Pseudupeneus grandisquamis,* Mullidae) and silver stardrum (*Stellifer illecebrosus*, Sciaenidae) (Figure 3.1). Both species inhabit sandy and muddy bottoms along the Eastern Pacific Ocean from Baja California, Mexico, to Peru (silver stardrum) and Chile (bigscale goatfish) (Chao 1995; Schneider 1995). Both exhibited prolonged periods of reproduction in the southern Gulf of California, with mature individuals found throughout the trawl season (September to March; Chapter 2). The majority of individuals of both species are immature when captured (Chapter 2). Chapter 2 focuses on elements of these species life histories contributing to biological productivity (i.e. survival, growth and reproduction).

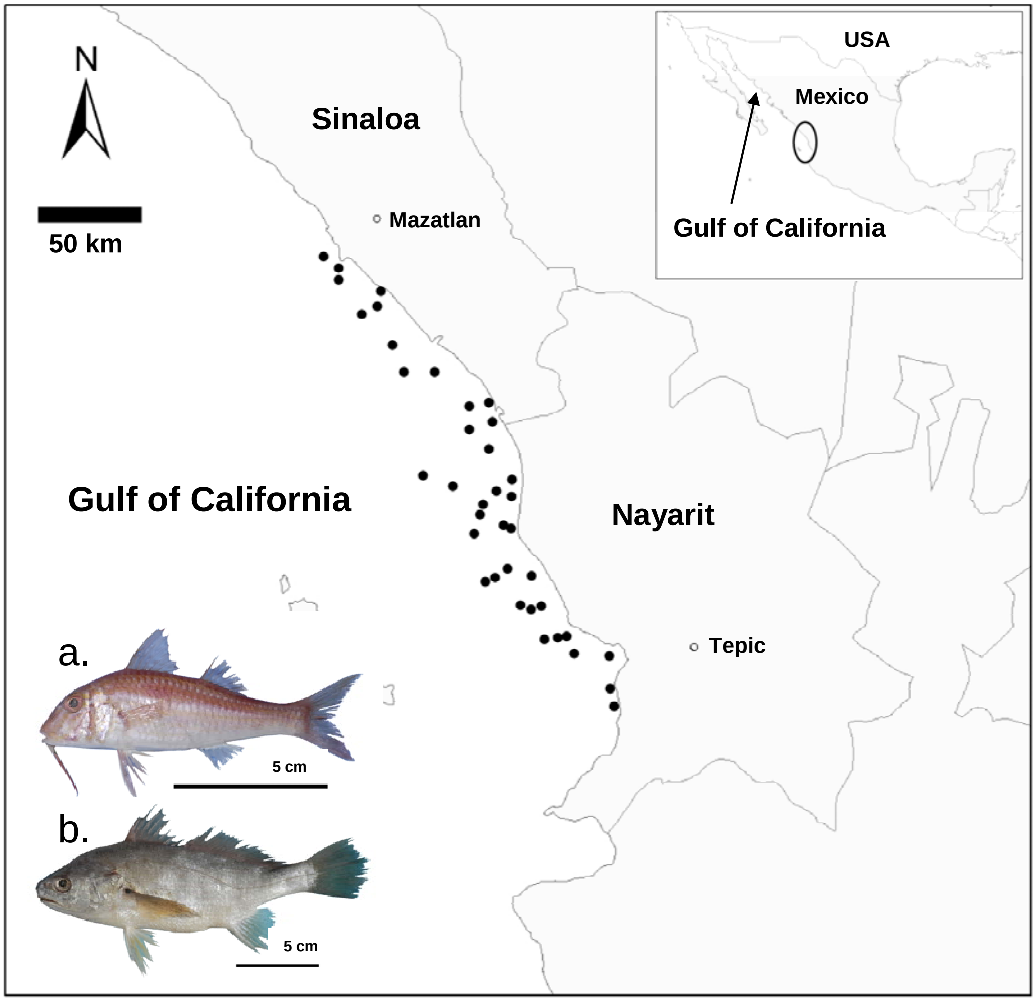

Figure 3.1. Map of Gulf of California, Mexico, indicating sampled area (inset), station locations and case study species a) bigscale goatfish, and b) silver stardrum. Species photos © CICIMAR.

Data collection

Information was collected on the location and timing of capture, as well as the quantity and size of the species when caught by the trawlers. Samples of bigscale goatfish and silver stardrum were collected and processed as per Chapter 2. In summary, a sample of the catch was collected from onboard two industrial shrimp trawlers, at 38 fixed stations across the shrimp fishing grounds of the southern Gulf. The stations were the same as those sampled annually by Mexican fisheries authorities to assess shrimp population status, and determine the opening dates for the fishery. One vessel sampled the stations from Nayarit (N = 21), the other from Sinaloa (N = 17). The two vessels, paired otter trawls – typical of the Gulf's industrial shrimp trawlers (see Gillet 2008 for description of gear) – followed the same sampling protocol to ensure comparability among all samples. The 38 stations were sampled monthly for the entire shrimp fishing season, from September 2006 to March 2007, except for those along the coast of Sinaloa where samples were not collected in November 2006 for logistical reasons (N = 17), and a few others, also for logistical reasons (N = 4). Stations occurred across an area measuring 11 600 km^2, and ranged in depth from 7 to 62 m. All individuals of the case study species were extracted from the samples, and frozen for later processing. On land, individual fish, by species and station, were counted and measured (total length = L_T, mm).

Response and predictor variables

Response variables of interest for both species were occurrence (presence/absence), density and mean L_T. Density was used instead of counts because it accounts for variations in gear (net length), and effort (tow duration), during sampling. Density was calculated as the total number of individuals per kilometre per hour of trawling (n km^{-2} hr^{-1}). The Gulf's trawlers fish with two nets. Area swept by the gear was calculated as two times the area swept by one net, which approximates net length * 0.55 * distance trawled (J. T. Nieto, CICIMAR, pers. comm. N.B.: Net length * 0.55 approximates the width of the net opening). The distance trawled was obtained using a GPS.

Predictor variables included depth, latitude and time, as they are the most relevant to spatio-temporal management. Limiting the number of predictor variables also reduced collinearity between variables, and the possibility of model over-parameterisation. Depth was measured as mean depth of a tow, in meters (m). Latitude (degrees North) was that of the vessel's position at the beginning of the tow. Time was measured as a continuous variable ('day', where 0 = first sample day = 25 September 2006), and as a categorical variable ('period', where Period 1 = 25 September 2006 – 11 December 2006, and Period 2 = 14 January 2007 – 25 March 2007). Total fishing effort differed significantly between the two periods. Fishing occurred 24 hours a day during the first half of the fishing season (Period 1), and the majority of licensed boats were on the water. In the second half of the season (Period 2), fishing only occurred at night, and fewer boats participated. Analytically, binning samples into two periods rather than many months increased the sample size, thus decreasing variation and improving ability to test for statistical differences between time periods.

Analyses

Analyses were carried out to determine the effects of depth, latitude and time (and/or their interactions) on the dependant variables (presence/absence, density, L_T). Spatial and temporal autocorrelation are almost certain to exist in data obtained from fisheries dependent sampling, as researchers have little control over when and where sampling occurs. There are various ways to analyse pseudoreplicated data (Fortin et al. 2002; Dormann et al. 2007), including:

a) **Temporal autocorrelation:** Look for spatial patterns, controlling for time, by calculating mean values of replicate measures at the same location, and assessing the extent of spatial patterns.

b) **Spatial autocorrelation:** Examine how time affects the dependent variables by comparing samples that were collected within a similar location at different times.

c) **Look for spatial and temporal patterns**, by analysing all data with a mixed-effects model.

These methods vary in their ease of interpretation for biological meaning, which is essential in a management context. Data were thus analysed using all three methods, in order to see if the simple methods, *a* and *b*, suggested the same management advice as the more complicated method, *c*. Simplifying the data and carrying out analyses on means (options *a* and *b*), has the benefit of results with clear biological meaning, which are easy to interpret and present. But, the results of simplified analyses can only be applied to the samples on which they were based, and averaging data can result in the loss of important information. The results of mixed-effects models (option *c*), can be generalised to the entire population, have increased statistical power as a result of using all available data, and give a flexible framework for analysing data with different types of correlations (Ives & Zhu 2006). But, unlike options *a* and *b*, the analyses are complicated, and the biological significance of the results can be difficult to interpret and explain, potentially limiting their usefulness in a management context.

a) Patterns in space, controlling for time

Each station was sampled 5-6 times across the study period. The effects of time were controlled for by conducting analyses on the proportion of presences (# presences * # times station was sampled^{-1}), mean density, and L_T, within each station, averaged across the study period. Regression analyses of these variables against mean latitude and depth of tows across the study period, within each station, were undertaken. Non-normally distributed variables were transformed as appropriate: log(square root) transformations for density, and log and inverse transformations for L_T. The errors presented below for slope and intercepts are standard errors.

b) Patterns in time, controlling for space

To control for spatial pseudoreplication, the effects of time period on the proportion of presences, mean density and L_T within latitude and depth bins, of 0.5 and 10 m, respectively, was examined. This enabled observations of the effects of time (periods), while controlling for space. Dependent variables for each spatial bin were compared between periods using a general linear model (for presence/absence data), and t-tests (for density and size data). Non-normally distributed variables were transformed as appropriate: log(square root) transformations for density, and log transformations for L_T.

c) Mixed-effects models

Mixed models provide a flexible framework for analysing data with different types of correlations. It was assumed that stations were a random sample from the population, and so presence/absence, density, and mean L_T were modelled using linear mixed models. Analyses were conducting using software provided in the packages lme (continuous variables) and lmer (binary variables) within the statistical program R (R Development Core Team 2004). Fixed effects were latitude, depth, day and their interactions. The random effect was station, to control for the non-independence of repeated sampling of the same station. All of the models' assumptions for continuous variables, that the residuals be normal, independent, and identically distributed, were verified as per Pinheiro and Bates (2002). Non-normally distributed data were transformed: log(square root) for density, and log for L_T. Autocorrelation of model residuals was checked using the autocorrelation function (acf) and by plotting a variogram for each of the dependent variables (Pinheiro and Bates 2002). Where autocorrelation of the residuals was found, it was modelled with a simple moving average model. The model of presence/absence was fit by Laplace approximation, and models of density and L_T were fit using maximum likelihood. The final model was selected using a sequential procedure for model selection, whereby all covariates were included at first, and then possible models of fixed effects were compared using likelihood-ratio tests and model fit criteria (Rabe-Hesketh & Skrondal 2005). Candidate models were evaluated by ANOVA and Akaike's Information Criterion (AIC). In the results for mixed effects models (below), the errors for covariates are standard errors.

Results

Bigscale goatfish

In space, controlling for time

When controlling for time, depth but not latitude was a significant predictor of the occurrence of bigscale goatfish. Bigscale goatfish were found with similar frequency across latitudes (slope = 0.13 ± 0.07; int = -2.22 ± 1.54; F = 3.49; R^2 = 0.07, P = 0.07, DF = 47; Figure 3.2a), but more frequently in deeper waters (slope = 0.006 ± 0.003; int = 0.55 ± 0.08; F = 5.46; R^2 = 0.12, P = 0.03, DF = 40; Figure 3.2b). Visual inspection of Figure 3.2b suggested that depth may be more important in water shallower than 30 m. Indeed, a regression through these points alone increased the variation in occurrence explained by depth (R^2) to 0.49 (slope = 0.28 ± 0.01; int = 0.14 ± 0.12; F = 22.42; P < 0.0001, DF = 24; Figure 3.2b – dashed line). The maximum depth at which bigscale goatfish was collected was 59 m (but only two stations were deeper than this).

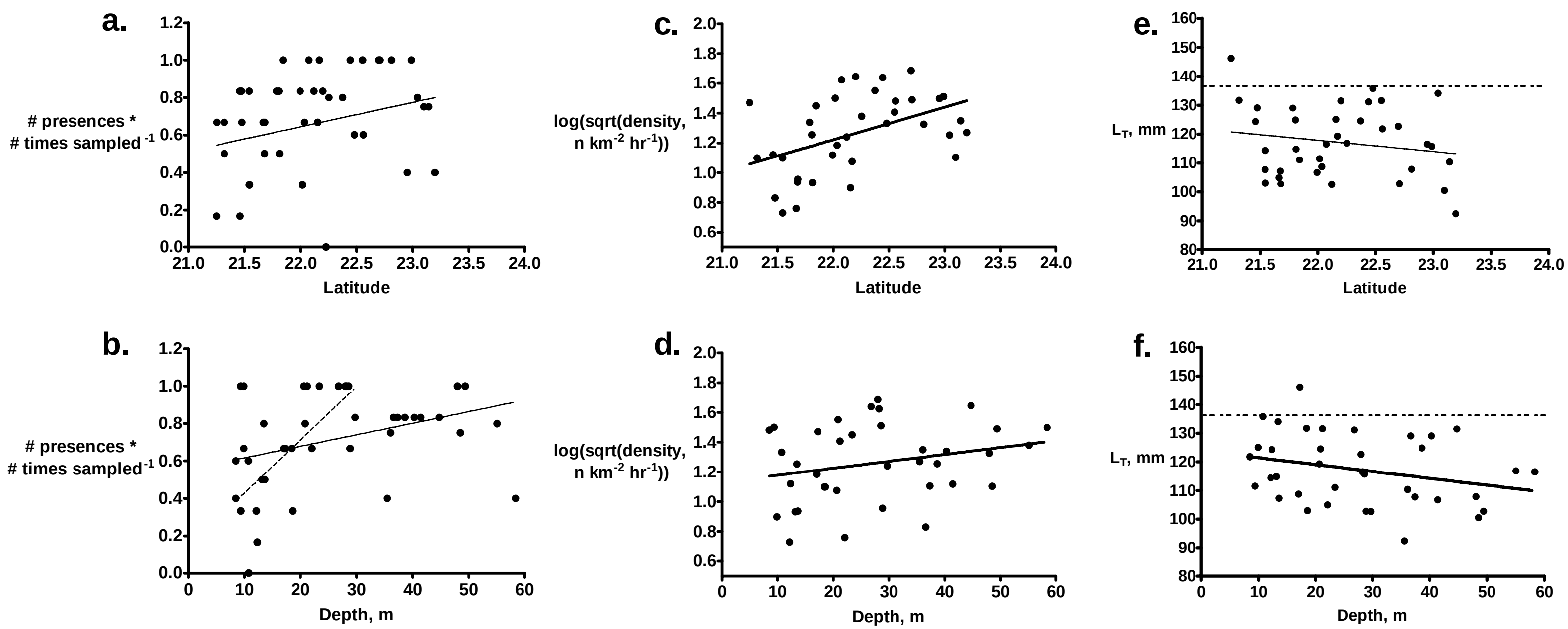

Figure 3.2. Analyses in space, controlling for time, for bigscale goatfish. Proportion of presences regressed against a) mean latitude (N = 48), and b) mean depth (N = 41), at each station. Dashed line in (b) is analysis on depths < 35 m (N = 25). Mean density (transformed) regressed against c) mean latitude (N = 37), and d) mean depth (N = 37), at each station. Mean total length (L_T) regressed against e) mean latitude (N = 37), and f) mean depth (N = 37), at each station. Dashed lines in (e) and (f) are at L_T at 50% maturity for this species (Chapter 2).

Density, on the other hand, was determined by latitude and not depth, though latitude only explained a small portion of variation in the data. The mean density (± standard deviation) of bigscale goatfish across all tows during the sampling period where this species was present was 1173 ± 2111 fish km^{-2} hr^{-1} (N = 145). Density was patchy as indicated by the large standard deviation to the mean. Density of bigscale goatfish increased with increasing latitude (slope = 0.22 ± 0.07; int = -3.58 ± 1.53; F = 10.06; R^2 = 0.22, P = 0.003, DF = 36; Figure 3.2c), but was more consistent across depths (slope = 0.005 ± 0.003; int = 1.13 ± 0.09; F = 2.31; R^2 = 0.06, P = 0.14, DF = 36; Figure 3.2d).

Neither of the spatial variables were significant determinants of size for bigscale goatfish, although there was evidence of decreasing size with increasing depth. Mean L_T (± standard deviation) of this species during the sampling period was 119 ± 24 mm (N = 6789). There was no relationship between density and mean size in the samples of bigscale goatfish (results not shown). Mean size of bigscale goatfish did not differ significantly across latitudes or depths (slope = -3.86 ± 3.58; int = 202.8 ± 79.37; F = 1.17; R^2 = 0.03, P = 0.29, DF = 36; slope = -0.24 ± 0.14; int = 123.80 ± 4.35; F = 2.90; R^2 = 0.08, P = 0.10, DF = 36, respectively; Figures 3.2e and f). While not significant, Figure 3.2f does suggest a negative trend of L_T with depth, such that smaller animals were found in deeper waters.

In time, controlling for space

Occurrence of bigscale goatfish increased in the second half of the sampling season (Period 2) at all latitudes, although only the increases in the southerly and mid-latitudes were significant, and in shallow and mid-depths (Figures 3.3a and b, Table 3.1). At greater depths, frequency of this species was not statistically significant between time periods (Figure 3.3b, Table 3.1). The analysis by depth also suggested that spatial patterns were not consistent between time periods. While bigscale goatfish appeared to occur more frequently in deeper waters in the first half of the fishing season (Period 1), this pattern was not apparent by the end of sampling (Period 2, Figure 3.3b).

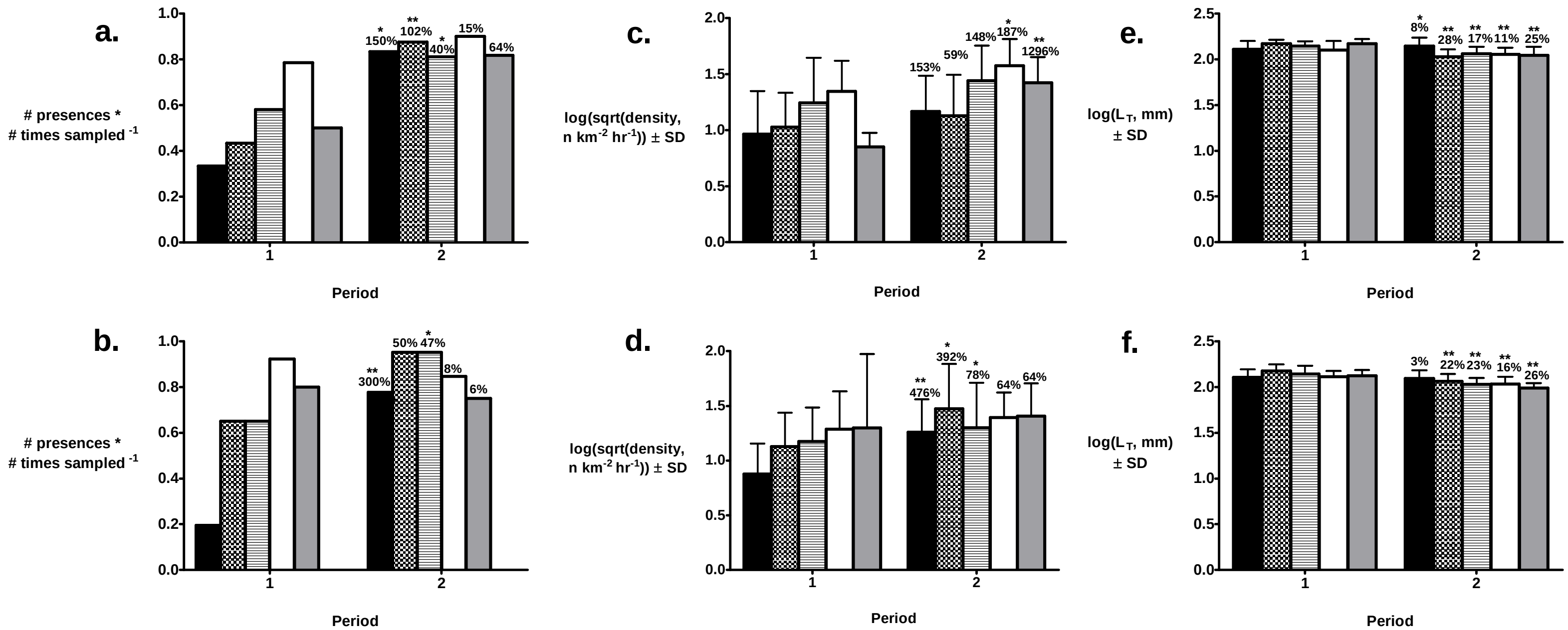

Figure 3.3. Analyses in time, controlling for space, for bigscale goatfish. Proportion of presences plotted against period in each of a) latitude, and b) depth, bins. Mean density (transformed) plotted against period in each of c) latitude, and d) depth, bins. Transformed mean total length (L_T) plotted against period in each of e) latitude, and f) depth, bins. Period 1 = 25 September 2006 – 11 December 2006, and Period 2 = 14 January 2007 – 25 March 2007. Percent change within bins (for raw data, not transformed) is indicated in Period 2. * P < 0.05; ** P < 0.01 (level of significance when corrected for multiple tests). LEGEND: ***Latitude bins in a, c and e:*** **dark = 21, hatched = 21.5, striped = 22, white = 22.4, grey = 23.** ***Depth bins (meters) in b, d, and f:*** **dark = 10, hatched = 20, striped = 30, white = 40, grey = 50. SD = standard deviation.**

Table 3.1. Analyses in time, controlling for space, for bigscale goatfish. Results of z-tests comparing presence and absences (PA), and t-tests comparing mean transformed density (log(sqrt(density); n km^{-2} hr^{-2}) and total length (log(L_T); mm) between time periods in several latitude (Northern) and depth (meters) bins. Period 1 = 25 September 2006 – 11 December 2006, Period 2 = 14 January 2007 – 25 March 2007. N_1, N_2 = tows for PA and density, individual fish for size.

	PA					DENSITY					L_T				
	N_1	N_2	DF	z	P	N_1	N_2	DF	t	P	N_1	N_2	DF	t	P
Latitude Bin															
21.0	12	12	23	2.33	0.02	4	10	12	1.01	0.33	40	218	256	2.10	0.04
21.5	28	32	59	3.61	<0.0001	13	28	39	0.87	0.39	150	791	939	20.78	<0.0001
22.0	31	37	67	2.03	0.04	18	30	46	1.88	0.07	530	2086	2614	23.65	<0.0001
22.5	14	20	33	0.91	0.36	11	18	27	2.39	0.02	386	2179	2560	11.14	<0.0001
23.0	8	11	18	1.42	0.15	4	9	11	4.59	0.001	9	412	419	4.01	<0.0001
Depth Bin															
10	36	45	80	4.84	<0.0001	7	35	40	3.06	0.004	42	983	1023	1.02	0.31
20	20	23	42	0.008	0.99	12	22	32	2.40	0.02	179	2585	2762	18.33	<0.0001
30	20	21	40	2.11	0.04	13	20	31	0.95	0.03	281	1204	1483	22.77	<0.0001
40	12	13	24	-0.53	0.59	12	11	21	0.87	0.39	487	507	992	17.31	<0.0001
50	5	8	12	-0.21	0.84	4	6	8	0.35	0.74	126	404	528	23.00	<0.0001

Mean densities of bigscale goatfish increased in the second half of the sampling season (Period 2) at higher latitudes (bins 22.5 and 23; Figure 3.3c, Table 3.1), and shallower depths (bins 10 and 20 m; Figure 3.3d, Table 3.1). The change was only significant in the most northern latitudes and shallowest depths once the level of significance had been corrected for multiple tests. The analysis re-enforces the spatial patterns observed in Figures 3.2a and b: density increasing with increasing latitude and consistent across depths.

Mean size of bigscale goatfish decreased in the second half of the sampling season (Period 2) across all latitudes, and at all depths but the shallowest (Figures 3.3e and f, Table 3.1). The possible stratification of size with depth observed in the spatial analysis (Figure 3.2f) is slightly visible in the second period, but not the first (Figure 3.3f).

Mixed effects model (all data)

Accounting for the random effect of station in a mixed effects model revealed that occurrence of bigscale goatfish increased with increasing depth (estimate = 0.17 ± 0.04; $z = 4.51$; $P < 0.0001$), and with the advancement of the fishing season (day: estimate = 0.05 ± 0.01, $z = 5.83$; $P < 0.0001$). While depth was more important than day in determining occurrence, their interaction was also important for this species (estimate = -1.20E-03 ± 3.00E-04; $z = -4.36$; $P < 0.0001$).

Mixed modelling also revealed that latitude and day were significant determinants of density for bigscale goatfish. Density increased with increasing latitude (estimate = 0.25 ± 0.07; $t = 3.32$; $P = 0.001$; DF = 106), and also as the fishing season progressed, though to a lesser degree (day: estimate = 1.50E-03 ± 4.00E-04; $t = 3.42$; $P = 0.001$; DF = 106).

Contrary to the spatial analysis, the mixed effects model, with log(mean L_T) per tow as the dependent variable, retained depth and day as important determinants of size for bigscale goatfish. Size decreased with increasing depth for this species (estimate = -1.90E-03 ± 4.00E-04; $t = -4.49$; $P < 0.0001$; DF = 108), and with the progression of the fishing season (day: estimate = -8.90E-04 ± 8.00E-05; $t = -10.65$; $P < 0.0001$; DF = 108).

Silver stardrum

In space, controlling for time

Silver stardrum were found more frequently in the southerly latitudes (slope = -0.42 ± 0.08; int = 7.79 ± 1.88; F = 14.76; R^2 = 0.29, P < 0.0001, DF = 37; Figure 3.4a) and in shallower waters (slope = -0.014 ± 0.003; int = 0.98 ± 0.10; F = 20.54; R^2 = 0.36, P < 0.0001, DF = 37; Figure 3.4b). Exceptions appeared to be the two most southerly stations, and removing these from the analysis increased the variation in occurrence explained by latitude (R^2) to 0.46 (slope = -0.43 ± 0.08; int = 10.19 ± 1.79; F = 28.88; P < 0.001, DF = 35; Figure 3.4a – dashed line). Visual inspection of Figure 3.4b suggested that depth may be more important in waters deeper than 30 m. Indeed, a regression through these points alone increased the variation in occurrence explained by depth (R^2) to 0.58 (slope = -0.027 ± 0.007; int = 1.55 ± 0.29; F = 16.37; P = 0.002, DF = 13; Figure 3.4b – dashed line), but above this threshold, latitude was the principle determinant of occurrence. The maximum depth at which silver stardrum was collected was 49 m (18 stations were deeper than this).

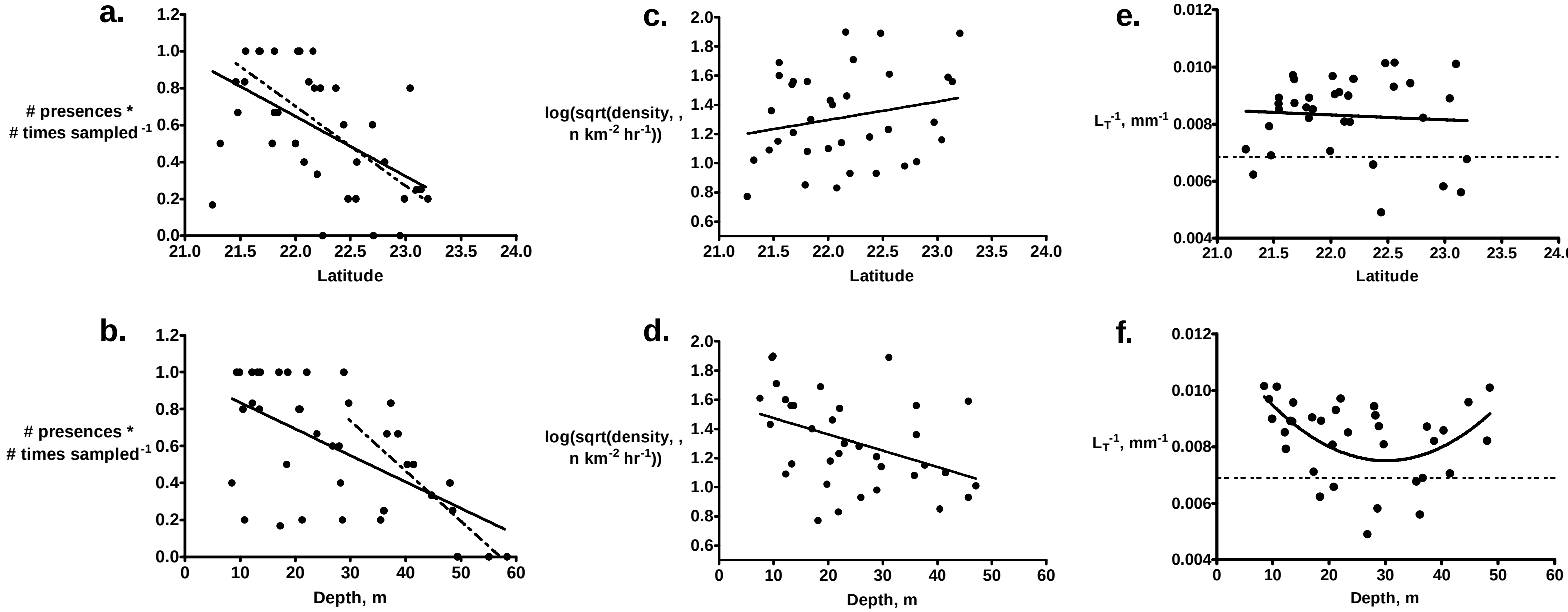

Figure 3.4. Analyses in space, controlling for time, for silver stardrum: Proportion of presences regressed against a) mean latitude (N = 38), and b) mean depth (N = 38), at each station. Dashed line in (a) is analysis without the two most southerly stations (N = 36), and in (b) is analysis on depths > 30 m (N = 14). Mean density (transformed) regressed against c) mean latitude (N = 35), and d) mean depth (N = 35), at each station. Transformed mean total length (L_T) regressed against e) mean latitude (N = 34), and f) mean depth (N = 34), at each station. Dashed lines in (e) and (f) are at L_T at 50% maturity for this species (Chapter 2).

Densities of silver stardrum decreased with decreasing depth. The mean density (± standard deviation) of silver stardrum across all tows during the sampling period where this species was present was 1444 ± 2045 n km^{-2} hr^{-1} (N = 126). Density was patchy as indicated by the large standard deviation to the mean. Density of silver stardrum was not related to latitude (slope = 0.13 ± 0.10; int = -1.47 ± 2.15; F = 1.68; R^2 = 0.05, P = 0.21, DF = 34; Figure 3.4c), but decreased significantly with increasing depth (slope = -0.011 ± 0.004; int = 1.59 ± 0.12; F = 6.77; R^2 = 0.17, P = 0.01, DF = 34; Figure 3.4d).

Mean L_T (± standard deviation) of silver stardrum during the sampling period was 113 ± 31 mm (N = 5634). The higher the density of silver stardrum at a station, the smaller the mean size (results not shown), suggesting that while larger individuals are spatially separated, smaller individuals are spatially aggregated. Mean size of silver stardrum was not related to latitude (slope = -1.73E-04 ± 4.30E-04; int = 0.01 ± 0.01; F = 0.16; R^2 = 0.01, P = 0.69, DF = 33, Figure 3.4e), but had a negative quadratic relationship with depth ($depth^2$ = 4.88E-06 ± 1.64E-06; depth = -2.93E-04 ± 9.20E-05; int = 0.012 ± 0.001; F = 5.42; R^2 = 0.26 P = 0.01, DF = 31, Figure 3.4f), such that the largest individuals were found at mid-depths (a quadratic function was a better fit to the data than a linear one, as determine by AIC). Analysis was conducted on $1/L_T$, as raw L_T data were highly right skewed, thus greater numbers on the y-axis of Figure 3.4f are smaller individuals.

In time, controlling for space

The frequency of occurrence of silver stardrum decreased in the second half of the fishing season in all latitude and depth bins, although only the changes in latitude bin 22.5, and at mid-depths (bins 20 and 30 m) were significant (Figures 3.5a and b, Table 3.2). Figures 3.5a and b appear to re-enforce observed spatial patterns (Figures 3.4a and b); a greater frequency of occurrence at lower latitudes, and a decrease in frequency of occurrence with increasing depth – patterns that were consistent between periods.

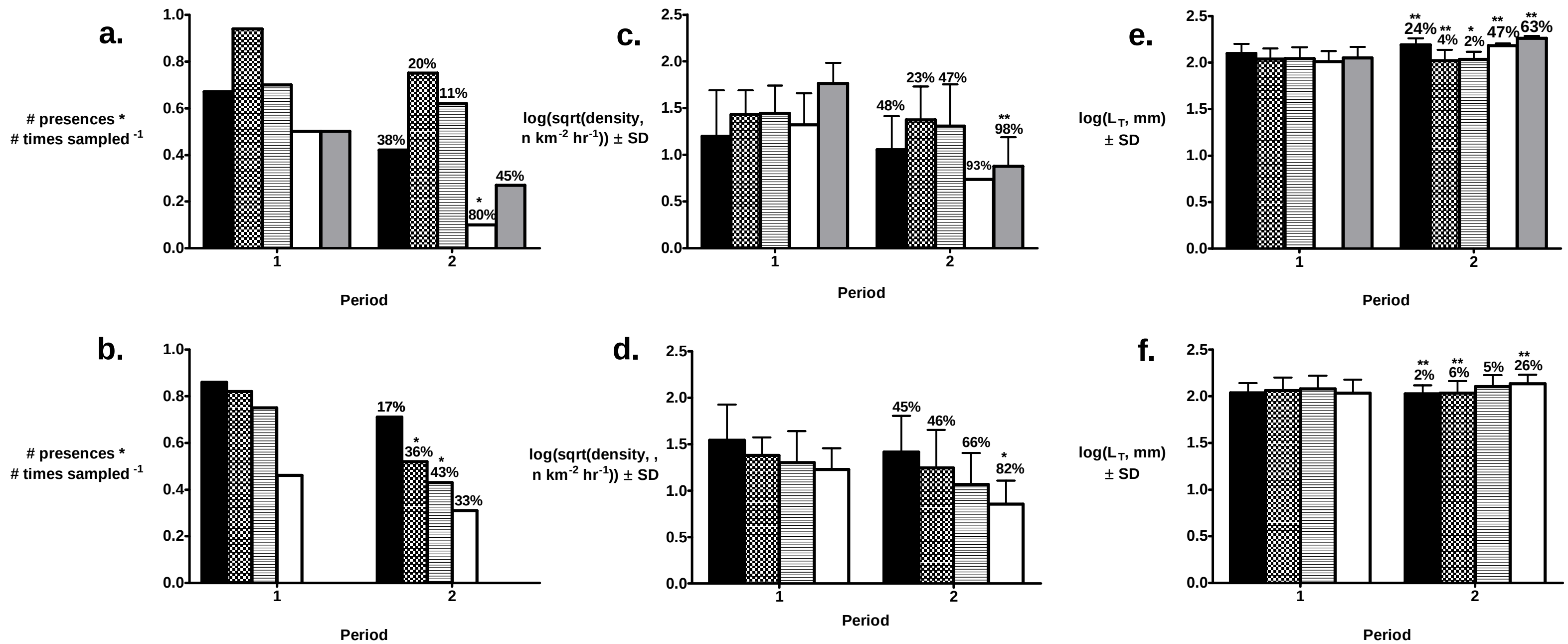

Figure 3.5. Analyses in time, controlling for space, for silver stardrum. Proportion of presences plotted against period in each of a) latitude, and b) depth, bins. Mean density (transformed) plotted against period in each of c) latitude, and d) depth, bins. Transformed mean total length (L_T) plotted against period in each of c) latitude, and d) depth, bins. Period 1 = 25 September 2006 – 11 December 2006, and Period 2 = 14 January 2007 – 25 March 2007. Percent change within bins (for raw data, not transformed) is indicated in Period 2. * P < 0.05; ** P < 0.01 (level of significance when corrected for multiple tests). LEGEND: *Latitude bins in a, c and e:* dark = 21, hatched = 21.5, striped = 22, white = 22.4, grey = 23. *Depth bins (m) in b, d, and f:* dark = 10, hatched = 20, striped = 30, white = 40. SD = standard deviation.

Table 3.2. Analyses in time, controlling for space, for silver stardrum. Results of z-tests comparing presence and absences (PA), and t-tests comparing mean transformed density (log(sqrt(density); n km^{-2} hr^{-2}) and total length (log(L_T); mm) between time periods in several latitude (Northern) and depth (meters) bins. Period 1 = 25 September 2006 – 11 December 2006, Period 2 = 14 January 2007 – 25 March 2007. N_1, N_2 = tows for PA and density, individual fish for size.

	PA					DENSITY					L_T				
	N_1	N_2	DF	z	P	N_1	N_2	DF	t	P	N_1	N_2	DF	t	P
Latitude Bin															
21.0	23	23	23	-1.22	0.22	7	6	11	0.58	0.57	139	104	241	8.03	<0.0001
21.5	32	32	62	-1.88	0.06	19	34	51	0.62	0.54	1373	1231	2602	3.87	<0.0001
22.0	30	37	66	-0.67	0.50	17	27	42	1.13	0.26	988	1467	2453	2.24	0.03
22.5	14	20	33	-2.40	0.02	7	2	7	2.33	0.05	89	5	92	3.34	0.001
23.0	8	11	18	-1.00	0.32	4	3	5	4.43	0.007	221	17	236	7.27	<0.0001
Depth Bin															
10	35	45	79	-1.53	0.13	22	40	60	1.25	0.22	2060	1989	4047	2.72	0.007
20	22	23	44	-2.05	0.04	15	15	28	1.15	0.26	445	559	1002	3.00	0.003
30	20	21	40	-2.04	0.04	12	12	22	1.68	0.12	249	257	504	1.78	0.08
40	13	13	25	-0.80	0.42	5	5	8	2.44	0.04	56	19	73	2.85	0.006

Density appeared to decrease in the second half of the fishing season (Period 2) in the higher latitude bins (22.5 and 23), however, only the change at the most northerly latitudes was significant (Figure 3.5c, Table 3.2). Some of the largest observed densities of silver stardrum occurred at the very lowest, and highest, latitudes at the start of the fishing season – but then densities at these stations declined, and were low to none for the rest of sampling. While density appeared to have decreased between periods at all depths (Figure 3.5d), only the difference at the deepest depths was statistically significant (Table 3.2).

Mean size increased significantly at all latitudes between periods (though the change in bin 22 was not significant after correcting for multiple tests; Figure 3.4e, Table 3.2). Changes in mean size with depth were less consistent. Mean size decreased slightly but significantly during the second half of the sampling season in depth bins 10 and 20 m, and increased slightly but significantly between seasons in depth bin 40 m (Figure 3.4f, Table 3.2).

Mixed effects model (all data)

Accounting for the random effect of station revealed that all three predictor variables, latitude, depth and day, were important determinants of the presence of silver stardrum across the study area. Again, spatial variables were more important than time. Occurrence decreased with increasing latitude (estimate = -1.48 ± 0.56; z = -2.67; P = 0.008), increasing depth (estimate = -0.09 ± 0.02; z = -4.06; P < 0.0001), and also occurred less frequently as the fishing season progressed (day; estimate = -0.015 ± 0.003; z = -4.54; P < 0.0001).

Mixed models also revealed that depth and day were important determinants of density for silver stardrum. Depth was the most important covariate, with density decreasing with increasing depth (estimate = -0.013 ± 0.003; t = -3.65; P < 0.0001; DF = 89). Density also decreased as the fishing season progressed (day; estimate = -0.0012 ± 0.0004; t = -2.94; P = 0.004; DF = 89).

Finally, a mixed model, with log(mean L_T) per tow as the dependent variable, agreed with the spatial analysis in that size was best predicted by a quadratic relationship with depth (depth = 0.010 ± 0.004; t = -2.53; P = 0.01; DF = 92; $depth^2$ = -1.64E-04 ± 7.73E-05; t = -2.12, P = 0.04;

DF = 92). Unlike all previous analyses, day was not an important predictor of size in this model, suggesting size of silver stardrum was stable over the sampling period.

Discussion

The findings in this paper confirm the difficulties of regulating shrimp trawl fisheries for particular bycatch species of concern, in that their life history and/or distribution commonly vary in a way that precludes general remedial action. Even the two species analysed here – a tiny sample of the many small species obtained as bycatch in Mexican shrimp trawls – exhibited distribution patterns that were at odds with one another. Were the species of equal conservation concern, then it would be hard to identify narrow spatial or temporal management measures to support both. In this case, it seems possible that fishing was exerting a lower toll on bigscale goatfish than on silver stardrum: juveniles of the former recruit to the study area over the fishing season despite the ongoing trawl fishery while the occurrence and density of the latter declined across the season. The two species did share a common trend that spatial variables mattered more than temporal ones, possibly arguing that permanent trawl free areas, and not temporal ones, might be required to mitigate potential trawl impacts on these fishes. After accounting for all small fish species, however, there is probably a strong argument for an extensive array of permanent trawl free areas, covering a range of depths and latitudes.

These two species really did differ in their temporal and spatial distribution. Latitude was an important predictor of presence for silver stardrum but not bigscale goatfish. While silver stardrum preferred shallow waters, bigscale goatfish preferred deeper waters. Occurrence of bigscale goatfish increased as the fishing season progressed across the study site, but decreased for silver stardrum. Latitude was significantly related to density of bigscale goatfish but not silver stardrum, whereas the reverse was true for depth. Densities of bigscale goatfish increased over the fishing season, whereas densities of silver stardrum decreased. Depth was a significant predictor of size for silver stardrum, and also bigscale goatfish, but to a lesser degree. And whereas mean size of silver stardrum may have increased over the sampling period, a pattern that was evident between periods, but not in the mixed model, mean size of bigscale goatfish decreased as sampling progressed.

The one consistent trend suggests that permanent trawl closures, and not temporal ones, would be required to mitigate potential trawl impacts on these fishes: spatial variables mattered more than temporal ones. This was evidenced in the results of the mixed effects models. For both bigscale goatfish and silver stardrum, depth and/or latitude mattered more than sample day in predicting occurrence, density and size.

The results also suggest that in addition to being permanent, trawl closures would need to be spatially diverse to protect even one of the case study species. Both species showed some heterogeneity of population structure across the study site, with observed distribution of size by depth. Such patterns are common to marine organisms, and need to be addressed in management measures (Field et al. 2006). Gradual ontogenetic movement to greater depths is common for shelf and slope species, and thus it was predicted that larger animals of both species would be found in deeper waters. The pattern for bigscale goatfish was opposite to that expected, with larger individuals in shallower waters. This could reflect larger, mature individuals returning to shallow waters to spawn. Sizes of silver stardrum were significantly related to depth, but the relationship was such that larger animals were found at mid-depths, and small animals in shallow and deeper waters. Further research is needed to understand the underlying processes that produce these patterns, but ideally a portion of all subsets of a species population would be protected by any trawl closures.

Bigscale goatfish may be recruiting into the study area during the sample period – such that juveniles are moving into new areas, and increasing in number in the areas they already occurred. This species occurred more frequently in the second half of the fishing season in many spatial bins, and density was greater in the second period in the more northerly latitudes and deeper waters. Observed increases in occurrence and density were accompanied by decreases in mean size in most spatial bins. Indeed, the proportion of immature individuals increased in the second half of the fishing season across the study site (Chapter 2). Bigscale goatfish have prolonged periods of reproduction (Ramos-Santiago et al. 2006; Chapter 2), with a potential reproductive peak in summer months (Ramos-Santiago et al. 2006). It is therefore possible that individuals spawned in summer months slowly recruit to the study site over the course of the year, with peak recruitment occurring approximately nine months after spawning.

There was no evidence for recruitment of silver stardrum to the study site, but the results do suggest potential for fisheries impact on this species. Silver stardrum decreased in occurrence at mid-depths and northerly latitudes over the course of sampling. Results of the mixed model suggested densities of silver stardrum decreased over the course of the fishing season. Observed decreases in occurrence and density may be the result of fishing pressure, although it is not possible to rule out the possibility of out-migration with our data. Any maintenance of density over the course of sampling could be explained by sites experiencing differing degrees of fishing pressure, or range collapse (such that individuals in less dense areas congregate to maintain a preferred density). Fisheries independent research and/or long-term data would be required to confirm the potential fisheries effects suggested by this study. Although there was no evidence of recruitment, silver stardrum also have prolonged periods of reproduction (Chapter 2). The proportion of mature individuals in the catch was more or less stable over the fishing season for this species (Chapter 2).

The fact that silver stardrum preferred shallow waters may also suggest that trawl fisheries have a greater impact on this species, compared to bigscale goatfish. While the industrial trawl fishery is closed from April to August, small scale boats trawl year round (García-Caudillo & Gómez-Palafóx 2005). These smaller boats cannot fish as deep as the industrial ones, and are thus more likely to affect shallow water species. This suggests potential for year round trawl effort being exerted on silver stardrum, whereas bigscale goatfish might get reprieve from trawl fishing in the closed season. Other fisheries operate in deeper waters during the summer months, but they do not involve bottom trawling. To fully understand potential impact of the fishery on these species one would need to overlay spatio-temporal distributions of fishing effort on these species distribution data (Andrew & Pepperell 1992), and/or monitor trends across several fishing seasons.

The inherit variability in natural systems, and the inability to control for the majority of such variability in observational studies, resulted in low correlation values for several of the significant analyses in space, controlling for time. Although low correlation values mean limitations in the explanatory power of relationships, the results are still useful in providing preliminary guidance for spatio-temporal management of these small fishes, and reflect relationships worthy of further investigation.

Excluded from the analyses was a physical variable, temperature, which may actually be influential in spatial distribution. The reasoning was that temperature was unlikely to be useful for the implementation of spatio-temporal management. Temperature was, however, predictably correlated to each of latitude, depth and time. Latitude was not an important determinant of occurrence for bigscale goatfish, which preferred deeper waters. But southerly latitudes did apparently favour silver stardrum, which are found in shallower waters, and thus may be subject to more critical temperature thresholds.

This research validates the utility of simple methods of analysis, which produced results very similar to those from mixed modelling. In most cases all analyses (simple or mixed models) drew similar conclusions regarding the spatio-temporal distribution of the case study species. One inference is that the research did not, therefore, suffer from the trap of autocorrelations (e.g. Hinch et al. 1994; Nash et al. 1999). The second inference is that the simple methods for dealing with temporal and spatial autocorrelation were sufficient for the purpose of understanding the case study species distributions. Similarly, another study found that correcting for spatial autocorrelation in cowbird distribution data improved predictive power, but that the simple models performed similarly and adequately overall (Jewell et al. 2007).

The decision between simple and complicated methods of analysis depends on whether one is looking for distribution patterns, or predictive equations. In three of the analyses completed here, the mixed models had the power to pick up effects that were too subtle to be detected by the simple approaches, but in all cases the important patterns were already evident in figures. First, while the regression analysis suggested that depth was not a significant determinant of size for bigscale goatfish, a decrease of size with increasing depth was suggested by the figure, and the mixed model agreed, retaining depth as a significant predictor of size for this species. Second, while the simple analyses suggested a non-significant decrease in density of silver stardrum between time periods, decreases were evident in the figures, and the mixed model retained day as an important determinant of density of this species. Third, while statistical analysis suggested significant changes in size of silver stardrum between the two time periods, the figures suggested such changes were minimal, and indeed the mixed model did not retain day as a determinant of size for this species. Such a discrepancy may also result from binning data into time periods which resulted in a loss of information that was retained by the mixed model.

In summary, the results of this study suggest that any trawl closures set in place to mitigate potential fishing effects on small fishes from their incidental capture in shrimp trawls would need to be permanent and biophysically diverse. Such a proposal might seem problematic were it not that most bottom trawling is unsustainable in its direct and indirect effects on even target species, large bycatch species, and marine habitats (e.g. Kaiser & Groot 2000; Gillett 2008). Closures would have obvious benefits for most taxa, especially where they occur (as in the Gulf of California shrimp fishery) in a context of declining target resources and increasing overheads (e.g. burdensome fuel costs; Gillett 2008; Sumaila et al. 2008).

Literature cited

Alverson, D. L., M. H. Freeberg, J. G. Pope, and S. A. Murawski. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper No. 339: 1-233.

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

Broadhurst, M. K. 2000. Modifications to reduce bycatch in prawn trawls: A review and framework for development. Reviews in Fish Biology and Fisheries **10**: 27-60.

Brusca, R. C., L. T. Findley, P. A. Hastings, M. E. Hendrickx, J. T. Cosio, and A. M. van der Heiden. 2005. Macrofaunal diversity in the Gulf of California. Pages 179-202 in J.-L. E. Cartron, G. Ceballos, and R. S. Felger, editors. Biodiversity, Ecosystems, and Conservation in Northern Mexico. Oxford University Press, Inc., New York, New York.

Bublitz, C. G. 1995. Mesh size and shape: Reducing the capture of undersized fish. Pages 95-99. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Chao, L. N. 1995. Sciaenidae. Corvinas, barbiches, bombaches, corvinatas, corvinetas, corvinillas, lambes, pescadillas, roncachos, verrugatos. Pages 1427-1518 in W. Fischer, F. Krupp, W. Schneider, C. Sommer, K. E. Carpenter, and V. Niem, editors. Guía FAO para Identificación de Especies para lo Fines de la Pesca. Pacifico Centro-Oriental. FAO, Rome.

Clucas, I. 1997. A study of the options for the utilization of bycatch and discards from marine capture fisheries. Page 59. FAO Fisheries Circular No. 928. Food and Agriculture Organisation of the United Nations, Rome.

Cochrane, K. L., editor. 2002. A fishery manager's guidebook - Management measures and their application. Food and Agriculture Organisation of the United Nations (FAO), Rome.

Delgado, C. L., N. Wada, M. W. Rosegrant, S. Meijer, and M. Ahmed 2003. Outlook for fish to 2020: meeting global demand. International Food Policy Research Institute and World Fish Centre, Washington, Penang.

Dormann, C. F., J. M. McPherson, M. B. Araújo, R. Bivand, J. Bolliger, G. Carl, R. G. Davies, A. Hirzel, W. Jetz, W. D. Kissling, I. Kühn, R. Ohlemüller, P. R. Peres-Neto, B. Reineking, B. Schröder, F. M. Schurr, and R. Wilson. 2007. Methods to account for spatial autocorrelation in the analysis of species distributional data: a review. Ecography **30**: 609-628.

Eayrs, S. 2007. A Guide to Bycatch Reduction in Tropical Shrimp-Trawl Fisheries. FAO, Rome.

Field, J. C., A. E. Punt, R. D. Methot, and C. J. Thomson. 2006. Does MPA mean 'Major Problem for Assessments'? Considering the consequences of place-based management systems. Fish and Fisheries **7**: 284-302.

Fortin, M.-J., M. R. T. Dale, and J. ver Hoef. 2002. Spatial analysis in ecology. Pages 2051–2058 in A. H. El-Shaarawi, and W. W. Piegorsch, editors. Encyclopedia of Environmetrics. John Wiley & Sons, Ltd., Chichester.

García-Caudillo, J. M., and J. V. Gómez-Palafóx 2005. La pesca industrial de camarón en el Golfo de California: Situación económico-financiera e impactos socio-ambientales. Conservación Internacional-Región Golfo de California, Guaymas.

Gillett, R. 2008. Global study of shrimp fisheries. Food and Agriculture Organization of the United Nations, Rome.

Hall, M. A. 1996. On bycatches. Reviews in Fish Biology and Fisheries **6**: 319-352.

Hinch, S. G., K. M. Somers, and N. C. Collins. 1994. Spatial autocorrelation and assessment of habitat-abundance relationships in littoral zone fish. Canadian Journal of Fisheries and Aquatic Sciences **51**: 701-712.

Ives, A. R., and J. Zhu. 2006. Statistics for correlated data: Phylogenies, space and time. Ecological Applications **16**: 20-32.

Jewell, K. J., P. Arcese, and S. E. Gergel. 2007. Robust predictions of species distribution: Spatial habitat models for a brood parasite. Biological Conservation **104**: 259-272.

Kaiser, M. J. 2000. The implications of the effects of fishing on non-target species and habitats. Pages 383-392 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Kaiser, M. J., K. R. Clarke, H. Hinz, M. C. Austen, P. J. Somerfield, and I. Karakassis. 2006. Global analysis of response and recovery of benthic biota to fishing. Marine Ecology Progress Series **311**: 1-14.

Kaiser, M. J., and S. J. Groot, editors. 2000. Effects of Fishing on Non-Target Species and Habitats. Blackwell Science, Ltd., Oxford.

Kelleher, K. 2004. Discards in the world's marine fisheries: An update. FAO, Rome.

Loverich, G. F. 1995. Thinking beyond the traditional codends. Pages 101-106. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Lozano-Montes, H. M., T. J. Pitcher, and N. Haggen. 2008. Shifting environmental and cognitive baselines in the upper Gulf of California. Frontiers in Ecology and Environment **6**: 75-80.

Morgan, L. E., and R. Chuenpagdee 2003. Shifting gears: addressing the collateral impacts of fishing methods in US waters. Island Press, Washington, DC.

Nash, C. H., J. S. Richardson, and S. Hinch, G. 1999. Spatial autocorrelation and fish production in freshwaters: a comment on Randall et al. (1995). Canadian Journal of Fisheries and Aquatic Sciences **56**: 1696-1699.

Pinheiro, J. C., and D. M. Bates 2002. Mixed Effects Models in S and S-Plus. Springer, New York.

R Development Core Team. 2004. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rabe-Hesketh, S., and A. Skrondal. 2005. Generalised Linear Mixed Models in B. S. Everitt, and D. C. Howell, editors. Encyclopedia of Statistics in Behavioural Sciences. John Wiley & Sons, Ltd., Chichester, U.K.

Ramos-Santiago, E., J. M. Ramirez-Gutierrez, R. Mendoza-Rodriguez, and M. Tapia-Garcia. 2006. Reproducción, distribución y abundancia del pez *Pseudupeneus grandisquamis* (Perciformes: Mullidae), en el Golfo de Tehuantepec, México. Revista de Biologia Tropical **54**: 1103-1112.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and R. Enríquez-Andrade. 2005. Rapidly shifting environmental baselines among fishers of the Gulf of California. Proceedings of the Royal Society of London B **272**: 1957-1962.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and J. P. Hawkins. 2006. The value of evidence about past abundance: marine fauna of the Gulf of California through the eyes of 16th to 19th century travellers. Fish and Fisheries **7**: 128-146.

Schneider, M. 1995. Mullidae. Salmonetes. Pages 1299-1300 in W. Fischer, F. Krupp, W. Schneider, C. Sommer, K. E. Carpenter, and V. Niem, editors. Guía FAO para Identificación de Especies para lo Fines de la Pesca. Pacifico Centro-Oriental. FAO, Rome.

Sherman, K., M. P. Sissenwine, V. Christensen, A. Duda, G. Hempel, C. Ibe, P. S. Levin, D. Lluch-Belda, G. Matishov, J. McGlade, M. O'Toole, S. Seitzinger, R. Serra, H. R. Skjoldal, Q. Tang, J. Thulin, V. Vanderweerd, and K. Zwanenburg. 2005. A global movement toward an ecosystem approach to management of marine resources. Marine Ecology Progress Series **300**: 241-296.

Stefansson, G., and A. A. Rosenberg. 2005. Combining control measures for more effective management of fisheries under uncertainty: quotas, effort limitation and protected areas. Philosophical Transactions of the Royal Society B **360**: 133-146.

Stockwell, D. R. B., and A. T. Peterson. 2002. Effects of sample size on accuracy of species distribution models. Ecological Modelling **148**: 1-13.

Sumaila, U. R., L. Teh, R. Watson, P. Tyedmers, and D. Pauly. 2008. Fuel price increase, subsidies, overcapacity, and resource sustainability. ICES Journal of Marine Science **65**: 832-840.

Walker, B. 1960. The distribution and affinities of the marine fish fauna of the Gulf of California. Systematic Zoology **9**: 123-133.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life-history diversification in North-American Fishes: Implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences **49**: 2196-2218.

4. Tropical shrimp trawl fisheries: Fishers' knowledge of and attitudes about a doomed fishery [3]

[3] A version of this chapter has been accepted for publication in Marine Policy. Foster, S.J. and A.C.J. Vincent. Tropical shrimp trawl fisheries: Fishers' knowledge of and attitudes about a doomed fishery.

Introduction

Tropical shrimp trawl fisheries are currently unsustainable in economic, environmental, and social terms. Economically, declining catch per unit effort (CPUE) combined with rising overhead costs (mainly fuel) and falling shrimp prices (due to world wide competition with lower-cost farmed shrimp) have reduced profitability for most of the worlds commercial shrimp trawl fisheries (Gillett 2008). As a consequence, half of all shrimp landings presently come from countries that subsidise fuel for such fisheries (Sumaila et al. 2008). Environmentally, the gear used in most large-scale operations is known to be destructive to marine habitats (Thrush et al. 1998; Kaiser 2000), as well as highly non-selective, catching many million tonnes of non-target species each year (Alverson et al. 1994). In the tropics, this bycatch is composed of (i) species of conservation concern, (ii) commercially important species targeted by other fisheries, and (iii) oft-forgotten small fish of considerable ecological, thought not economic, value (Alverson et al. 1994; Kelleher 2004). Socially, conflict among the industrial shrimp fishing sector and other, smaller-scale, fishing sectors is a common concern (e.g. FAO 2001; Gillett 2008). The conflicts mainly arise where the industrial shrimp fishery catches species of economic importance to other fisheries and/or displaces fishers that use static gears.

Tropical shrimp trawl fisheries around the world employ similar sets of mitigation measures to address the direct and indirect costs of their practices. The most common management measures to improve the status of shrimp stocks include controlling fishing effort through permit requirements, vessel buy-back programs, mesh size regulations, and closed seasons and/or areas (Gillett 2008). In addressing the indirect impacts of shrimp trawling, managers commonly try to reduce overall fishing effort, promote modifications to fishing gear – mainly through the use of turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) – and ban fishing in areas of critical habitat or species of conservation concern (Hall & Mainprize 2005). Although many of these technical measures are useful (Broadhurst 2000; Eayrs 2007), trawl closures are likely to be most effective in reducing fishing mortality and habitat impacts (Eayrs 2007). Retention of bycatch, often promoted as a mitigation measure for tropical shrimp trawl fisheries, does not reduce environmental impact (Davies et al. 2009). Social concerns can, however, be reduced by moving larger boats offshore, in order to reduce physical conflict with smaller-scale operations (Gillett 2008).

Successful implementation of any management or mitigation measure requires information. An understanding of effort distribution in space and time is needed to plan for spatial/temporal management, but also to understand sector overlap and so potential for fisheries conflicts. Assessing BRDs for effectiveness requires an understanding of relative values of different components of the catch, including bycatch. Although gear modifications and changes to fishing techniques have proven successful at reducing bycatch in some fisheries, failure to consider the regulatory and social systems in which the gear modifications are being implemented has resulted in considerable resistance to their introduction (Bache 2003).

Little is known about shrimp fishers' knowledge and attitudes, despite the large number of scientific papers, technical manuals and management briefs on tropical shrimp trawl fisheries, especially with respect to bycatch (e.g. Vendeville 1990; Robins et al. 1999; Stobutzki et al. 2000; FAO 2001; Eayrs 2007; Gillett 2008). The available literature considers the biological elements of managing tropical shrimp trawl fisheries but pays little attention to social acceptance of such protocols, despite the clear importance of human behaviour in determining the success of fisheries management. As just one example, the US government mandate to use TEDs and BRDs, although biologically reasonable, met considerable resistance from shrimp trawl fishers in the Gulf of Mexico because they perceived a loss of independence and control over the circumstances of their work (Johnson et al. 1998).

Fishers' knowledge can be useful in both biological and management contexts (e.g. Scholz et al. 2004). They know a lot about marine life because their livelihoods depend on maximizing catch while minimizing effort (Mackinson & Nøttestad 1998). Tapping into stakeholder knowledge should increase greater acceptability of the results of the research, and resulting recommendations (Mackinson & Nøttestad 1998). In a review of marine conservation planning initiatives, those that were most successful seemed to include more stakeholder groups (Leslie 2005). In spite of this, past practices in fisheries management have generally involved only biologists and economists, attention turning to fishers' perceptions only in the last two decades or so (Couper & Smith 1997).

The industrial shrimp trawl fishery in the Gulf of California, Mexico, provides a useful model for the challenges of managing tropical shrimp trawl fisheries, not least because of its great economic and social importance (García-Caudillo & Gómez-Palafóx 2005). Despite its prominence, the fishery suffers the same problems as other tropical shrimp exploitation. Reported management issues include declining catches and overcapacity, unprofitability (government subsidies are presently needed to maintain an otherwise unviable fishery), conflict between it and small-scale fisheries, and environmental impacts (Young & Romero 1979; Perez-Mellado & Findley 1985; García-Caudillo & Gómez-Palafóx 2005; Gillett 2008). Current management includes a seasonal closure (April-September, to protect commercial shrimp species), permit requirements, depth restrictions (< 10 meters), area closures (bays, estuaries, and a few marine protected areas (MPAs)) and gear requirements (minimum mesh size and TED) (Meltzer & Chang 2006). Enforcement of fishing laws is the responsibility of federal government through CONAPESCA, with little room for local governments to manage fisheries resources (Gillett 2008).

Ours is the first study on industrial shrimp trawl fishers' perceptions and/or attitudes in the Gulf, and complements previous studies on the direct and indirect issues associated with the fishery (e.g. Young & Romero 1979; Arreguín-Sánchez et al. 2002; García-Caudillo & Gómez-Palafóx 2005), and the technical solutions to these problems (e.g. García-Caudillo et al. 2000). In this paper, we use interviews with fishers from the industrial shrimp trawl fishery of the Gulf of California to shed light on the social dimensions of tropical shrimp fisheries management. Specifically, we wished to obtain fishers' views on direct and indirect problems facing the fishery, proposed and potential management options to address the issues, and the future of the fishery in general. We also wanted to obtain new knowledge on effort and valuable components of bycatch to feed into the management process. We focus our work on fishers from the southern Gulf of California, the area of the Gulf with greatest intensity of anthropogenic pressure, and greatest increasing trend of such pressure (Enríquez-Andrade et al. 2005).

Methods

Interviews

We interviewed shrimp trawl fishers from the two main fishing ports in the southern Gulf of California: Mazatlan, Sinaloa and San Blas, Nayarit (Figure 4.1). Limiting interviews to the southern Gulf (also known as Lower Gulf), one of the three biogeographic regions (Walker 1960), avoided results being influenced by biogeographic differences in species composition and abundance. Interviews took place in March 2006 (Mazatlan only) and January-March 2007 (Mazatlan and San Blas). We intentionally directed our interviews at more experienced fishers, in a non-random fashion. Selection of fishers was done either by word of mouth, asking one candidate to suggest other experienced fishers to talk to (i.e. snowball sampling, Neis et al. 1999; Huntington 2000), or by direct interception (Richardson et al. 2005). Where we used the interception method to find fishers, we sought out individuals whom we guessed to be older.

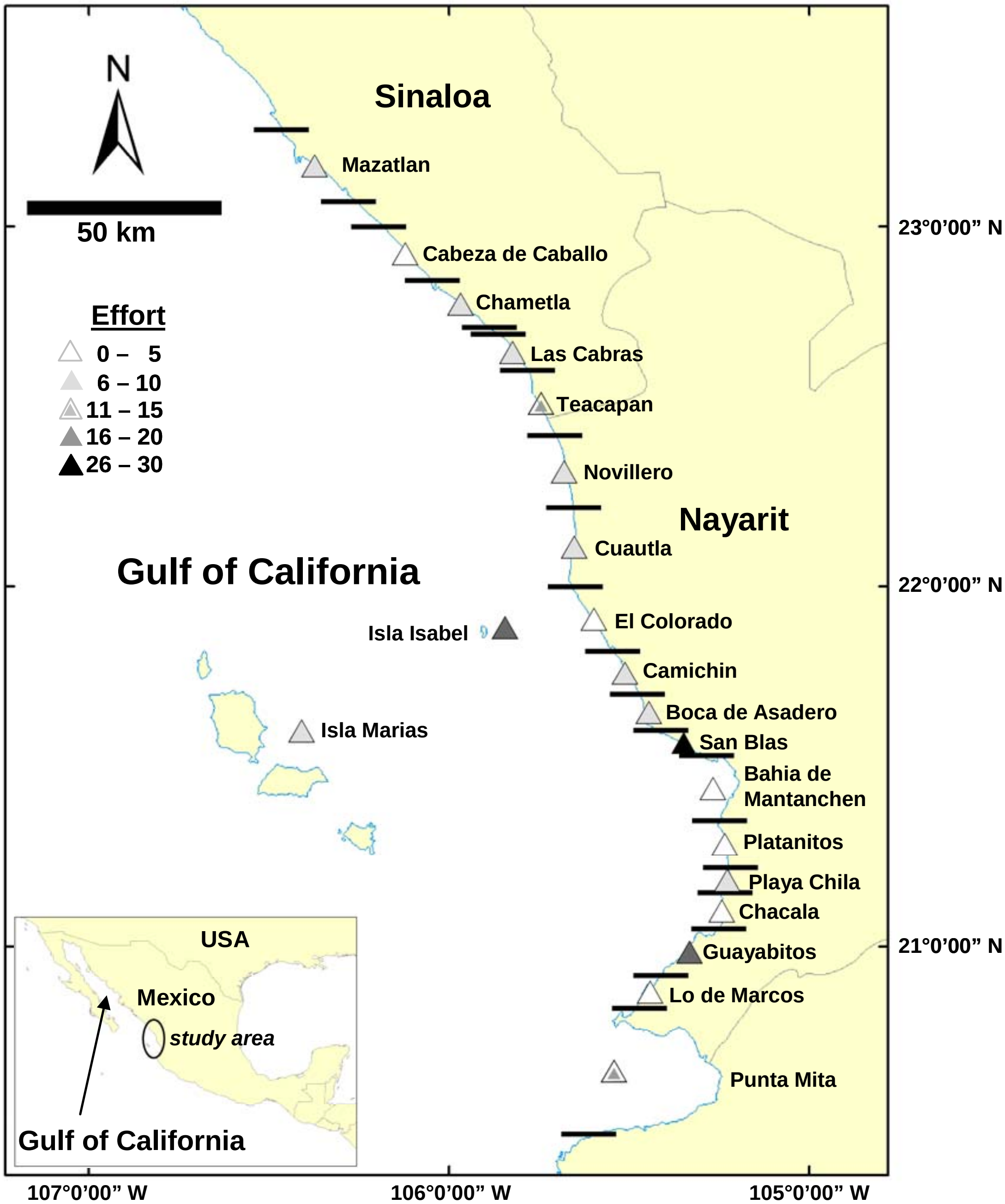

Figure 4.1. Industrial shrimp trawl effort across fishing grounds of the southern Gulf of California, Mexico. Relative effort was determined from interviews with fishers from the ports of Mazatlan, Sinaloa and San Blas, Nayarit (N = 46), and was measured as the number of respondents to report an area as having a high concentration of trawlers, year to year. Bars along coast delineate fishing areas.

Interviews followed a semi-structured multi-purpose questionnaire that was designed to yield information on the fishery, catches, changes over the years the fishery had been operating (in catches and fishing practices), issues facing the fishery and potential options for addressing the problems. During the interviews, participants were guided in the discussion by the interviewer based on a predetermined list of questions (Table 4.1), but the direction and scope of the interview were allowed to follow the participant's train of thought. This provided opportunity for novel information to come up in the conversation (Huntington 2000). As a result of this flexibility, the number of responses varied for each question (as indicated in the results), and the interviews varied in length (although most were around 30 minutes). Interviews were conducted by S.J.F. with the help of a local Mexican research assistant. S.J.F. relied on the assistant for interpretation in the first year and then drew on him more for validation during the second year. Responses were noted but not tape recorded. This research was given ethics clearance by The University of British Columbia.

Table 4.1. Questions used to guide semi-structured interviews with industrial shrimp trawl fishers from the southern Gulf of California, Mexico.

1. Information on fishery:
a. Can you identify areas you believe experience a high concentration of fishing boats, every year?
b. Has your fishing practice changed over time?

2. Information on bycatch:
a. Does bycatch affect the location of your fishing?
b. Do you keep any of the bycatch? Which species?

3. Changes in catch over time
a. Has the total weight of your catch changed over time? Please quantify.
b. Have shrimp catches changed over time? Please quantify.
c. Has the amount of fauna in the catches changed over time? Please quantify.
d. Has the amount of valuable fishes in the catches changed over time? Please quantify.
e. Have there been any changes in the specific fish species you catch over time?

4. Issues with the fishery (asked in 2007 only)
a. In your opinion are the shrimp stocks you catch stable, decreasing, or increasing?
b. What do think might be the primary cause(s) for your observations?
c. How do you see the future of the shrimp trawl fishery in Mexico?

5. Possible solutions (asked in 2007 only)
a. What solutions do you propose for the problems facing the shrimp trawl fishery?
b. What do you think about the following possible solutions:
  Bycatch excluder devices
  Reducing the number of boats
  Reducing the length of the season
  Closing areas to trawling

We asked fishers who reported changes in catch rates (increasing or decreasing) to quantify the change. We then converted their responses to % change per year in order to make individual responses comparable. When fishers did not provide a time frame for a reported change, we used the number of years the fisher had participated in the fishery as the maximum time frame over which the change had occurred. This would result in conservative decline rates if fishers were actually reporting declines that had occurred over a shorter period of time than their time in the fishery. All statistical analyses were conducted with GraphPad Prism, version 5.02 (GraphPad Software Inc., 2008).

During interviews, fishers identified taxa by their local common names. The local names were then translated to scientific names by the research assistant. The final list was verified by an independent researcher, a trained taxonomist with extensive experience working with bycatch from the shrimp trawlers in our study area. Most reported declines in specific taxa were not quantified by fishers. Where fishers did quantify declines, the units were variable such that we could not compare, nor deduce, mean rates, and thus we present individual fisher comments. We should note that fish catches are variable in space and time, so when fishers reported catching tonnes of big fish per tow they would not mean every tow, but when they happened to catch them.

Respondents

We interviewed a total of 52 fishers for the study: 20 fishers in Mazatlan in 2006; 17 fishers in Mazatlan in 2007; and 15 fishers in San Blas in 2007. Fishers worked aboard 34 different trawl vessels: we interviewed one fisher from each of 25 trawlers, two fishers from each of seven trawlers, six fishers from one trawler, and seven from another. All fishers were interviewed alone, including those that worked on the same boat. The respondents and vessels represented in this study are only a small fraction of those found in the southern Gulf. In 2006, the states of Sinaloa and Nayarit registered 767 and 20 vessels respectively (SAGARPA 2006), which with an average of six fishers per vessel, translates into approximately 4602 and 120 fishers in each respective state.

Interviewed fishers had spent a mean of 20 ± 11 years participating in the fishery (N = 52). The majority of the fishers could be considered experienced as the commercial shrimp trawl fishery in the Gulf only started 60 years ago; only seven had spent less than 10 years fishing, and 15 had spent 30 or more seasons on the boats. The majority of fishers interviewed were captains, mechanics or sailors. The number of fishers interviewed by position, in order of rank importance, was (Spanish name in parentheses): captain (capitán, N = 17), mechanic (motorista, N = 15), cook (cocinero, N = 6), sailor (marinero, N = 12) and deck hand (pavo, N = 1).

The majority of respondents (N = 33) worked in other fisheries-related jobs during the closed season for shrimp trawling. The most common source of income during the closed season was a small-scale fishery called *escama* (scale), where fishers reported targeting *inter alia* barracuda, dorado, mackerel and shark (N = 15). Eight and four fishers were active in shark and tuna fishing, respectively. Two respondents worked as fisheries observers on tuna boats, and one respondent was active in each of the following fisheries related jobs: beach seine fisher, breath-hold diver (oysters and lobsters), buyer/seller, processor, tuna mariculture. Reported non-fisheries related jobs included maintenance of the shrimp trawlers (N = 4), farmer (N = 3), security guard (N = 2), carpenter, hotel cook, electrician, and mechanic (each with N = 1). The rest of the fishers claimed they only rested during the closed season (N = 6).

Results

The following sections synthesise responding fishers' views on direct and indirect problems facing the fishery, potential management options to address the issues, and the future of the fishery in general. We also present new knowledge on effort and valuable components of bycatch to feed into the management process. Fisher's views on problems and management options were either *prompted* by researchers during the interview, or volunteered (*unprompted*) by respondents during our discussions.

Direct problems

Prompted

Approximately half of fishers who commented on the status of commercial shrimp populations considered them stable (N = 13/27), while the other half considered them to have declined over time (N = 14/27). Unless otherwise stated, 'time' is defined throughout this chapter as the number of years a fisher had been a participant in the fishery, with a mean value of 20 ± 11 years across all respondents. None of the respondents thought the Gulf's shrimp populations had increased over their time in the fishery. In a clearer verdict, the majority of fishers who commented on shrimp yields had experienced a decline in catches over time (N = 31/35), with a mean reported decline rate of 4% ± 2% year^{-1} (N = 11). Very few respondents reported stable shrimp catches (N = 4/35), and none reported increases.

The majority of respondents reported declines in total catch (shrimp plus bycatch) over time (N = 24/29), while only five reported catches as stable, and none reported an increase. The mean reported decline in total catch was 4 ± 2% year^{-1} (N = 15). Seven fishers commented that the shrimp to bycatch ratio had not changed over time, supported by the fact that mean reported decline rates for total catch and shrimp catch were the same.

Unprompted

All 32 fishers interviewed in 2007 commented on problems (direct and indirect: Table 4.2), with individual fishers reporting an average of 4 ± 2 problems (range 1-7). The number of problems reported by an individual fisher was not related to their time in the fishery (linear regression: slope ± SE = -0.04 ± 0.03; int ± SE = 4.50 ± 0.62; F = 2.18; R^2 = 0.07; P = 0.15; DF = 31), nor their position within the fishery (ANOVA: F = 1.74; P = 0.18; DF = 31). Similarly, there was no apparent correlation with years nor position in the fishery and each problem individually (results not shown). The most commonly reported concern (89% of respondents) was too many industrial trawl boats, resulting in a decrease in catch per boat. This was followed by decreasing shrimp prices and increasing overhead costs, with 41 and 34% of fishers citing these issues, respectively (Table 4.2).

Table 4.2. Unprompted direct and indirect problems facing the industrial shrimp trawl fishery in the Gulf of California, Mexico, as volunteered by participating fishers (total N = 32).

ISSUE	# respondents	REASONS (based on fisher interviews)
DIRECT		
too many industrial trawl boats	29	has resulted in a decrease in catch per boat
declining shrimp prices	13	due to large amounts of shrimp entering the market from aquaculture and small-scale shrimp trawl fishing operations (*pangas*), and at half the price as those from industrial fishing operations
increasing overheads	11	due to the rising costs of oil and age of the fishing vessels, as old boats require more of maintenance and cost more to run that younger boats
bigger, more efficient, gears	4	heavier wood doors cause more damage to the bottom; more efficient gears catch more of everything
illegal mesh size	1	fishers used 1.5 inch mesh in the past, until the law mandated 2.25 inches, but now boats were again using 1.5 inches to catch as many shrimp as possible
improved technology	1	fishers can find the shrimp faster, and fish more of the Gulf than they used to
INDIRECT		
small-scale shrimp trawl fishers affect industrial shrimp catches	26	*pangas* impact the catches of industrial fishers because they fish illegally in the closed season and in shallow waters, and start fishing first (their season opens a couple of weeks before the industrial season)
poor governance	15	the government does a poor job enforcing laws, especially when it comes to small-scale shrimp trawl fishers who are considered to fish illegally in the closed season and areas without penalty
aquaculture	11	take larvae from the bays and estuaries, thereby reducing the number of mature shrimp available to industrial shrimp fishers
contamination/garbage	5	fishers throw garbage overboard, including engine oil and old nets
industrial shrimp fishers impact catches of (non-shrimp) small scale fishers	1	trawlers catch fishes that are important to small-scale fishers
climate change	1	climate change is changing currents and therefore locations of shrimp and fish

Indirect problems

Prompted

Approximately half of fishers who commented on trends in bycatch (the incidental portion of their catch) considered it stable (N = 18/32), while the other half considered it to have declined over the years they had participated in the fishery (‘time’) (N = 14/32). None of the respondents thought the amount of bycatch in their catches had increased over time. When fishers were asked to comment specifically on large fishes (we did not define large for them), almost all respondents reported declines in this part of the bycatch (N = 33/35). Only two reported large fish catches as stable, and none reported them as having increased over time. Several specific taxa were reported by fishers (unprompted by us) as having declined over their years participating in the fishery (Table 4.3).

Ten fishers commented on whether the amount or types of bycatch affected where they chose to fish. Of these the majority said they would avoid fishing an area if there was a lot of bycatch compared to shrimp (N = 6). Three respondents reported targeting large fishes at the end of the season when shrimp catches were low; one even commented that he used a larger mesh size at the end of the season so as to target larger fishes. Only two respondents claimed that the fauna does not affect where they fish, and one mentioned keeping the small fish component of the bycatch to sell for fish meal.

Table 4.3. Fish taxa reported by industrial shrimp trawl fishers from the southern Gulf of California, Mexico, as having declined over time. Fishers were asked to comment on observed declines in fishes in general, but specific fish taxa were unprompted.

Common name (Spanish)	N	Kept[a]	Family	Genus/species	Max size (cm)[b]	Habitat[b]	Form	Diet[b]	Common name (English)[b]	Fishers' comments[c]
Robalo	26	yes	Centropomidae	*Centropomus nigrescens, C. viridis*	123, 112	demersal	elongated and slender	zoobenthos (including shrimps), nekton	snook (Black/ White)	40 bags to less than 1 bag per trip (26 years); 300 kg to 1 individual per season (20 years); 750 kg first trip, now a few individuals all season (3 years)
Constantino	2	yes	Centropomidae	*Centropomus robalito*	35	pelagic-neritic	elongated and slender	zoobenthos (including shrimps), nekton	Yellowfin snook	
Pargo	24	yes	Lutjanidae	*Lutjanus colorado*	91	reef-associated	oblong, moderately laterally compressed	zoobenthos	Colorado snapper	70 to 9 individuals per trip (16 years); used to catch 6-10 kg individuals, but not anymore (25 years);
Huachinango	4	yes	Lutjanidae	*Lutjanus peru*	95	reef-associated	oblong, moderately laterally compressed	zoobenthos, nekton	Pacific red snapper	300 to 25 kg per season (10 years); could catch 7000 kg in 3 days, but not anymore (5 years)
Caballo del mar	10	?	Syngnathidae	*Hippocampus ingens*	31	demersal, reef-associated	swims upright	zooplankton	seahorse	
Mero	9	*yes*	Serranidae	*Epinephelus* spp.			oblong, moderately laterally compressed		grouper	caught 10-80 kg individuals in past, but now only 15 kg when catch at all (25 years)
Botete	9	yes	Tetraodontidae	*Sphoeroides annulatus*	44	demersal	round	zoobenthos, plants	Bullseye puffer	
Mojarra	7	yes	Gerreidae	*Eucinostomus* spp./ *Diapterus* spp.		demersal	laterally compressed		mojarra	would catch 300 kg in a two hour tow, now get 100 kg in a 4 hour tow (% years)

Common name (Spanish)	N	Kept[a]	Family	Genus/species	Max size (cm) [b]	Habitat[b]	Form	Diet[b]	Common name (English) [b]	Fishers' comments[c]
Burro	6	yes	Haemulidae	*Pomadasys panamensis*	39	demersal	oblong, moderately laterally compressed	zoobenthos, nekton	Panama grunt	500 to 45 kg per town (6 years)
Chivito	5	no	Mullidae	*Pseudupeneus grandisquamis*	30	demersal	flat on ventral surface, moderately laterally compressed	zoobenthos	Bigscale goatfish	
Chihüil	3	yes	Arridae	*Bagre pinnimaculatus*	95	demersal	robust, flat on ventral surface, spines	nekton	Red sea catfish	
Lenguado	3	yes	Paralichthyidae	*Cyclopsetta panamensis, C. querna*	35, 39	demersal	flattened dorso-ventrally	zoobenthos, nekton	God's, Toothed flounder	
Corvina, Corvina chana	3	yes	Sciaenidae	*Cynoscion phoxocephalus*, C. *stolzmanni*, *C. reticulatus*	60, 115, 90	demersal	flatter on dorsal surface, moderately laterally compressed	zoobenthos, nekton (including shrimps)	Cachema, Stolzmann's, Striped weakfish	
Baqueta	3	yes	Serranidae	*Epinephelus acanthistius*	100	demersal	oblong, moderately laterally compressed		Rooster hind	80 to 3 individuals per haul (35 years)
Chile	2	?	Synodontidae	*Synodus* spp.		demersal	elongated and rounded		lizardfish	
Tiburón*	2	yes	Sphyrnidae/ Carcharhinidae	*Sphyrna lewini/ Rhizoprionodon longurio*	430/ 110	pelagic-oceanic/ bentho-pelagic	elongated, thick	zoobenthos, nekton (including shrimps)	shark (Scalloped hammerhead/ Pacific sharpnose)	
Dorado	1	yes	Coryphaenidae	*Coryphaena hippurus*	210	pelagic-neritic	elongated, thick	nekton	Common dolphinfish	

Common name (Spanish)	N	Kept[a]	Family	Genus/species	Max size (cm)[b]	Habitat[b]	Form	Diet[b]	Common name (English)[b]	Fishers' comments[c]
Chabela	1	yes	Stromatidae	*Peprilus medius, P. snyderi*	25, 30	bentho-pelagic	compressed laterally		Pacific harvestfish, Salema butterfish	
Liceta	1	yes	Mugilidae	*Mugil curema*	90	reef-associated	elongated, rounded on ventral surface	zooplankton, zoobenthos, nekton, plants	White mullet	
Mantarraya	1	*yes*	Gymnuridae/ Dasyatidae	*Gymnura marmorata/ Dasyatis brevis, D. longus*	100/ 187, 260	demersal	flattened dorso-ventrally	zoobenthos, nekton	California butterfly ray/ Whiptail, Longtail stingrays	
Berugatta	1	yes	Sciaenidae	*Micropogonias altipinnis*	90	bentho-pelagic	oblong, rounded on dorsal surface	zoobenthos (including shrimp), nekton	Tallfin croaker	
Sierra	1	yes	Scombridae	*Scomberomorus sierra*	99	pelagic-neritic	elongated, moderately laterally compressed	zoobenthos, nekton	Pacific sierra	

[a] yes = mentioned by fishers in interviews as species they keep, *yes* = not mentioned in interviews but are know to be retained, and ? = unclear if retained.
[b] Data from references in Fishbase (Froese & Pauly 2009)
[c] Years in parentheses are either the number of years over which the decline had occurred as reported by the fisher, or when this information was missing, the number of years the fisher had participated in the fishery.

Unprompted

The most commonly reported indirect problem was small-scale shrimp trawl fishers (81%, Table 4.2). Respondents commented that human population growth along the coast had resulted in an increased number of small-scale shrimp fishers, many of which are in direct competition with industrial boats as they can fish the same depths (up to 40 m) and with similar gears (nets up to 18 m). Poor governance and aquaculture operations were the next most commonly reported problems, with 47% and 34% of responding fishers citing these issues, respectively (Table 4.2).

Management options

Prompted

Three-quarters of fishers who commented on trawl free zones (MPAs) considered them a good idea (77%), but only if adequate enforcement was put in place (Table 4.4). One respondent even remarked that MPAs would be respected only if the Mexican president himself was sitting in the middle. Those against MPAs did not perceive benefits from them. Two respondents commented that existing zones closed to trawling – bays and estuaries, and any water shallower than 10 meters – were trawled anyway.

Table 4.4. Industrial shrimp trawl fishers' views on potential management options for their fishery in the Gulf of California, Mexico. Management options were either prompted by the researcher, or volunteered by participating fishers during discussions (unprompted). N = number of respondents who commented on the management option; n = sample size where a further breakdown of overall result is reported.

MANAGEMENT OPTION	FOR	AGAINST	N	REASONS (based on fishers interviews)
PROMPTED				
MPAs	20	6	26	*for*: would only work with adequate enforcement *against*: would limit the availability of fishing grounds, further reducing shrimp yields
reduce fleet size	20	2	22	*for*: government buy-back program best way to achieve this (n = 10) *against*: would put people out of jobs
shorten trawl season	17	9	26	*for*: should end earlier (n = 8), start later (n = 4) *against*: would result in less employment (n = 6)
TEDs	7	21	28	*for*: does not affect catches *against*: affects shrimp catches, prevent large fish from entering the nets, are dangerous
UNPROMPTED				
better governance (increased vigilance)	16			government does not care about fisheries resources – evidenced by the fact they allowed small-scale fishers to trawl in the closed season, and inside estuaries
temporary trawl moratorium	6			average suggested length of 3.25 years, but government would have to help find alternative sources of income for displaced fishers – whether from other employment, or government assistance
spectra nets/ regulate mesh size	3			considered to catch less fauna and reduce fuel consumption
standardise gears	2			all boats have to be the same size, and use the same sized gears – thereby reducing competitive advantage of some larger boats
eliminate small-scale shrimp trawlers	2			
zoning of fishing grounds	2			fishers would be restricted to fishing in their home state
ban all trawling	1			

Almost all fishers who commented on reducing fleet size considered it a good idea (91%, Table 4.4). Three respondents commented that the fleet size was decreasing anyway, as old boats stopped working, but also as overhead costs rose and fishers could not afford to go out. However, one respondent commented that reducing fleet size would change the problem from an ecological one to a social one, as there would be people out of work. Another respondent commented that all trawlers should be eliminated, large and small scale, and be replaced with alternative fishing methods that are not as harmful to the environment.

Shortening the fishing season was supported by 65% of fishers who commented on this management option (Table 4.4). Most respondents felt it should close earlier, as shrimp yields are not profitable after January/February, but several others felt the season should start later, as shrimp populations are still reproducing and/or too small in September (Table 4.4). Two respondents suggested a mid season break, as they believe that shrimp have a second reproductive period around February. One respondent suggested spatio-temporal restrictions that were planned around target shrimp species. Fishers first target shallow and then deep water shrimp species during the first and second halves of the fishing season, respectively. The fisher proposed, therefore, that deeper waters be closed to fishing in the first half of the season, and shallow waters in the second half.

It was clear from the interviews that fishers considered TEDs a problem, and not a solution; 75% of fishers who commented on the device were against it (Table 4.4). The main reasons were that they (i) frequently get clogged (with garbage, sticks, mud etc.) and thus prevent shrimp from passing into the cod end, (ii) exclude large fish, which supplement fishers income, and (iii) are dangerous to handle due to their weight. Though we did not ask about problems and management options in 2006, dissatisfaction with the TED was the only unprompted issue discussed by respondents at that time (N = 7/20).

Unprompted

Nearly all fishers interviewed in 2007 commented on management options other than those prompted by the interviewer, with individual respondents suggesting an average of 2 ± 1 options (range 1-3) (Table 4.4). The number of management options suggested by an individual fisher was not related to their time in the fishery (linear regression: slope ± SE = -0.02 ± 0.01; int ± SE = 1.91 ± 0.23; F = 3.73; R^2 = 0.14; P = 0.07; DF = 24), nor their position within the fishery (ANOVA: F = 0.676; P = 0.58; DF = 24). Similarly, there was no apparent correlation with years or position in the fishery and each option individually (results not shown).

The most common unprompted management option was better governance, especially increased vigilance of the small-scale shrimp trawl fishers (64% of respondents, Table 4.4). Two respondents suggested the enforcement system to be corrupt, such that for a fee small-scale fishers are warned ahead of time that authorities are paying an enforcement visit. In San Blas only, a common unprompted management option was a temporary trawl moratorium (N = 6/15 San Blas fishers, Table 4.4). Two respondents, one from Mazatlan and one from San Blas, also suggested zoning of fishing grounds (Table 4.4).

Future of the fishery

Most fishers who commented felt that the fishery had no future (N = 27/30). Specific stated reasons for the eventual demise of the fishery included collapsed shrimp populations (N = 6), the destruction of the marine environment (N = 3), and a loss of economic viability as increasing overheads combine with decreasing shrimp prices (N = 7). Two of the three fishers who considered the fishery to have a future said it would continue only if the government started to take better care of fisheries resources, if no one fished when the shrimp were reproducing, and if the fleet size was maintained or (better) reduced.

Information to support management

Effort

Almost all respondents were willing and able to describe the main fishing grounds in the southern Gulf of California (between Mazatlan, Sinaloa and Cancun, Jalisco) (N = 46/52). A query as to where they fished elicited a usual response of “everywhere”. We therefore asked respondents where, year to year, they expected to find a large concentration of trawlers, and defined fishing effort as the number of individuals who cited the fishing ground as having a high concentration of trawlers (Figure 4.1).

About half of fishers interviewed commented on what makes a good fishing ground (N = 23). Spatial variables included areas in front of lagoons (N = 2) and rivers (N = 6). Temporal factors included temperature (N = 11), presence of a full moon (N = 6), and rain (N = 6).

Important bycatch species

Several bycatch species were reported as being retained by fishers (Table 4.3). These taxa range in maximum sizes from 25 to almost 500 cm, in habitat (although the majority are demersal), and in shape (from elongate and rounded to dorso-ventrally flattened) (Table 4.3). In addition, the majority of these taxa relied on the shrimp grounds for their food source, some including shrimps as a targeted part of their diets (Table 4.3).

Discussion

Our study reveals fishers' knowledge and attitudes that should be considered when developing management plans for industrial shrimp trawl fisheries. It is notable that among the problems facing the Gulf of California fishery, fishers tended to identify those generated externally – fluctuations in availability of the resource, increases in fishing effort, decreases in international price of shrimp and increases in operation costs – and thus distance themselves from responsibility for management options. In such a climate, any solutions to the fishery are likely to depend on proper enforcement and reliable governance, as our study indicates. Should strong enforcement be put in place, then trawl free areas seem to be the most pragmatic way to alleviate problems associated with the fishery; our effort data point to areas that might have greatest acceptance among fishers.

The Gulf's industrial trawl fishers agreed with official reports that the fishery is suffering overcapacity, which depletes both the targeted shrimps and the ecosystem in general. Our respondents and official statistics have reported declines in catch per vessel over time

(SAGARPA 2004), although fishers perceived the rate of decline to be more drastic (4% versus 1% per year, respectively, CONANP et al. 2004). A similar discrepancy in perceived decline rates was found in the northern Gulf, where fishers reported a much steeper decline in shrimp biomass than official records (Lozano-Montes et al. 2008). The fishers rightly blamed excessive fishing effort for decreased catches – the onset of declines in catch rates coincided with a significant increase in the number and size of fishing vessels (SAGARPA 2004). Fishers did not agree as to whether reduced shrimp catches also meant declines in shrimp biomass, though reports suggested that most shrimp stocks in the Gulf of California are fully or over-fished (e.g. SAGARPA 2000). Our study adds to existing anecdotal evidence of reductions in the Gulf's overall biomass (e.g. Sáenz-Arroyo et al. 2005b; Sáenz-Arroyo et al. 2006; Lozano-Montes et al. 2008).

In our interviews, respondents recognised that even if effort were reduced, and catch per vessel increased, shrimp trawl industries might still be unprofitable because of declining shrimp prices and increasing overheads. As industrial shrimp catches in the Gulf failed to keep up with demand, Mexico's shrimp aquaculture grew rapidly to compensate (García-Caudillo & Gómez-Palafóx 2005), such that current production is now almost equal to wild capture (Gillett 2008). Low prices for aquaculture shrimp and shrimp from small-scale fisheries have forced industrial shrimp trawling operations to sell their product at a loss (García-Caudillo & Gómez-Palafóx 2005). Declining shrimp prices combined with the rising price of oil and an ageing fleet – in 2006, 82% of the Gulf's vessels were older than 20 years (SAGARPA 2006) – means that the fleet is generally unprofitable, with 93% of vessels registering a loss in 2000 (García-Caudillo & Gómez-Palafóx 2005).

Fisher interviews were only somewhat useful for increasing our understanding of the indirect impacts of shrimp trawl fisheries, especially with respect to the issue of bycatch. Fishers suggested several larger fish species for which incidental capture in shrimp trawl nets might be a problem; some of which other Gulf fishers had previously identified as depleted through fishing (Sáenz-Arroyo et al. 2005b). It is notable that most fishers reported declines in total catch (bycatch plus shrimp) without any change in bycatch to shrimp ratio yet, oddly, failed to recognise that bycatch species must therefore have declined with shrimp; only half reported a decline in bycatch over time. Such discrepancies suggest that fishers' knowledge may be most confidently embraced for species that they value (as per Sáenz-Arroyo et al. 2005a; Sáenz-

Arroyo et al. 2005b; Lozano-Montes et al. 2008). The corollary is that alternate assessment methods will be needed for the small fish component of bycatch, which constitutes most incidental catch in shrimp trawl fisheries (Alverson et al. 1994).

Our study shows that a lack of understanding of the value fishers place on bycatch will limit the utility of bycatch reduction devices and other remedial action. Fishers value the very species which such devices serve to exclude. For example, the TEDs are unpopular because they allow mobile fish – of great value to the fishers – to escape along with the turtles. Ironically, such TEDs do little to exclude the small fish bycatch that fishers would be happy to reduce. The situation is similar in Indonesia, where compliance with the government mandated use of TEDs is very low, with the main reason being that they reduce fish bycatch which generate income higher than monthly wages (Zainudin et al. 2007). Studies on TED performance have, therefore, failed in focusing only on their effects on shrimp catch and not on mobile or larger fish. This challenge to BRDs emphasises the need for management ventures to consider socioeconomic impacts and not just ecological goals. In general, research needs to refocus on assessing impacts and limiting unsustainable bycatch rather than only eliminating certain components of it (Horsten & Kirkegaard 2002).

The fishers we interviewed placed most blame for the problems facing their fishery on sectors other than their own, suggesting low levels of ownership for the current situation. The fishers blamed small-scale shrimp trawl fishers, the government and aquaculture ventures. Placing the blame on the small scale shrimp trawl fishery suggests that industrial shrimp fishers will not likely respect management or mitigation measures directed at the industrial fleet until the government has regulated the small-scale fishery. Indeed, restructuring of the small-scale fishery has been identified as a necessary step towards improving the shrimp industry in Mexico (Meltzer & Chang 2006). Tightly tied to the issue of the small scale trawl fishery is that of poor governance. Small-scale fishers fish illegally, in both space and time, and some fishers we interviewed suggested this happens without penalty, even considering the fisheries management system in Mexico to be corrupt. Aquaculture operations also bore fishers' blame, as they are perceived to steal larvae and thus decrease the number of mature shrimps available to industrial fishers, although it is more likely that aquaculture affects the industry by lowering market prices for shrimp (as discussed above).

Although fishers indicated support for several commonly employed shrimp trawl fisheries management options, it is clear from our study that improved governance and effective enforcement are vital for such options to be successful. Fishers generally supported three of the common management measures employed in tropical shrimp fisheries worldwide, professing willingness to reduce effort (a) spatially with trawl free zones, (b) overall by reducing fleet size, and (c) temporally with a limited fishing season. Regardless, the perceived lack of effective governance means that even mitigation measures supported by fishers would fall short of achieving their goals. A review of the world's shrimp fisheries suggest management is complex where it involves small-scale fishers, is open access, or occurs in a poor country with weak institutional arrangements for management (Gillett 2008). The Gulf's shrimp fishery suffers all three of these, and all three need to be addressed before hope for a better future is returned to its fishers. Though revealed to be an important issue for the fishers, poor governance is not explicitly mentioned in existing literature addressing the management of the Gulf's shrimp fishery (e.g. Meltzer & Chang 2006; Gillett 2008).

Of the three proposed options for reducing overall trawl effort, we suggest that trawl free areas are the most pragmatic way forward with shrimp fisheries management in the Gulf of California (and indeed shrimp fisheries in general). First, the number of fishing vessels is likely to decline without intervention, as the boats age and become prohibitively costly to run. Second, shortening the fishing season would have relatively little impact on catch (including bycatch), given diminishing returns with time (García-Caudillo & Gómez-Palafóx 2005). And third, unless fishers have other sustainable ways to earn an income, reductions in shrimp exploitation (through a reduction in fishing vessels, or a shortening of the fishing season) would probably just redirect fishing effort onto other marine resources (Cheung & Sumaila 2008).

Trawl free areas would probably achieve levels of protection far beyond those of any BRD (Eayrs 2007). They would mitigate many of both the direct and indirect issues associated with the fishery, and fishers would generally support them as long as the government enforces them. Moreover, it is unlikely that any BRD or other technical solution will provide effective support for the bycatch species, given their very diverse sizes, shapes and habitats (Bublitz 1995; Loverich 1995). There are presently 11 MPAs in the Gulf, some of which have displaced trawlers, but none has been put in place as an explicit part of the shrimp fishery management plan. Instead, most were aimed at protecting charismatic species or specific high-profile areas

(Enríquez-Andrade et al. 2005), or managing and enhancing small-scale fisheries while conserving marine ecosystems (Cudney Bueno et al. 2009). The success of the restricted areas will depend on proper enforcement – it is helpful that electronic vessel monitoring (VMS) is already in place for all the Gulf's industrial trawlers (Gillett 2008) – and on exclusion of small-scale fishers.

Our effort data, while limited, shed light on the potential placement of trawl free areas in the southern Gulf of California. Although VMS monitoring of all industrial shrimp trawlers has been underway since 2004, the information gathered is not made readily available, even to in-country researchers. The effort data we obtained through fisher interviews is, therefore, important despite its uncertain and patchy nature. We were able to get the names of the fishing grounds between Mazatlan and Puerto Vallarta and an indication of which may be more important to fishers, although all information could do with further verification. In the meantime, our data may help to place trawl free zones where they might be accepted. For example, our findings suggest that trawl free areas should not be placed in front of lagoons and river outputs, areas identified by fishers as important for shrimp populations.

Our study revealed that fishers lack hope for a better future for the industrial shrimp trawl fishery in the Gulf of California. Indeed, decreasing shrimp prices and increasing overheads may lead to eventual economic extinction of the fishery, even if shrimp stocks remain stable. It is estimated that these fisheries are highly subsidised, with a cost of 1.6 pesos to generate one peso of income from shrimp trawling in the Gulf (García-Caudillo & Gómez-Palafóx 2005). This social cost is likely to have increased since time of publication given subsequent decreases in shrimp prices and the rising price of oil. Elimination of subsidies could reduce capacity by eliminating older vessels that operate at a loss each year (approximately 600 boats, and 3500 to 4000 jobs; García-Caudillo & Gómez-Palafóx 2005). A reduction in capacity would clearly complement marine zoning for trawl free areas. In the long run, however, it may be decreasing shrimp prices and increasing operation costs that drive the fishery to economic extinction, and so reduce pressure on bycatch species.

Literature cited

Alverson, D. L., M. H. Freeberg, J. G. Pope, and S. A. Murawski. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper No. 339: 1-233.

Arreguín-Sánchez, F., E. Arcos, and E. A. Chávez. 2002. Flows of biomass and structure in an exploited benthic ecosystem in the Gulf of California, Mexico. Ecological Modelling **156**: 167-183.

Bache, S. J. 2003. Bycatch mitigation tools: selecting fisheries, setting limits, and modifying gear. Ocean and Coastal Management **46**: 103-125.

Broadhurst, M. K. 2000. Modifications to reduce bycatch in prawn trawls: A review and framework for development. Reviews in Fish Biology and Fisheries **10**: 27-60.

Bublitz, C. G. 1995. Mesh size and shape: Reducing the capture of undersized fish. Pages 95-99. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Cheung, W. L., and U. R. Sumaila. 2008. Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. Ecological Economics **66**: 193-210.

CONANP, CONAPESCA, CI, IN, ENOS, and SEMARNAT. 2004. Pesca sustentable de camarón en el Golfo de California: Propuesta de reestructuración de la flota camaronera de altamar. http://www.conservacion.org.mx/conservacion_internacional/pdf/propuesta%20reestructuraci%F3n.pdf.

Couper, A. D., and H. D. Smith. 1997. The development of fishermen-based policies. Marine Policy **21**: 111-119.

Cudney Bueno, R., L. Bourillon, A. Saenz-Arroyo, J. Torre-Cosio, P. J. Turk Boyer, and W. W. Shaw. 2009. Governance and effects of marine reserves in the Gulf of California, Mexico. Ocean & Coastal Management **52**: 207-218.

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

Eayrs, S. 2007. A Guide to Bycatch Reduction in Tropical Shrimp-Trawl Fisheries. FAO, Rome.

Enríquez-Andrade, R., G. Anaya-Reyna, J. C. Barrera-Guevara, M. d. l. Á. Carvajal-Moreno, M. E. Martínez-Delgado, J. Vaca-Rodríguez, and C. Valdés-Casillas. 2005. An analysis of critical areas for biodiversity conservation in the Gulf of California region. Ocean and Coastal Management **48**: 31-50.

FAO 2001. Tropical shrimp fisheries and their impact on living resources. Shrimp fisheries in Asia: Bangladesh, Indonesia and the Philippines; in the Near East: Bahrain and Iran; in Africa: Cameroon, Nigeria and the United Republic of Tanzania; in Latin America:

Colombia, Costa Rica, Cuba, Trinidad and Tobago, and Venezuela. FAO Fisheries Circular, No. 974. Food and Agriculture Organisation, Rome.

Froese, R., and D. Pauly, editors. 2009. FishBase - World Wide Web electronic publication, version (03/2009).

García-Caudillo, J. M., M. A. Cisneros-Mata, and A. Balmori-Ramírez. 2000. Performance of a bycatch reduction device in the shrimp fishery of the Gulf of California, Mexico. Biological Conservation **92**: 199-205.

García-Caudillo, J. M., and J. V. Gómez-Palafóx 2005. La pesca industrial de camarón en el Golfo de California: Situación económico-financiera e impactos socio-ambientales. Conservación Internacional-Región Golfo de California, Guaymas.

Gillett, R. 2008. Global study of shrimp fisheries. Food and Agriculture Organization of the United Nations, Rome.

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

Horsten, M. B., and E. Kirkegaard. 2002. Bycatch from a perspective of sustainable use. Page 16. IUCN SSC European Sustainable Use Specialist Group, Brussels.

Huntington, H. P. 2000. Using traditional knowledge in science: methods and applications. Ecological Applications **10**: 1270-1274.

Johnson, G. D., C. Formichella, J. S. Thomas, D. Bhaumik, F. V. I. Degruy, and C. A. Riordan. 1998. Stress and distress among Gulf of Mexico shrimp fishermen. Human Organisation **57**: 404-413.

Kaiser, M. J. 2000. The implications of the effects of fishing on non-target species and habitats. Pages 383-392 in M. J. Kaiser, and S. J. de Groot, editors. Effects of fishing on non-target species and habitats: Biological, conservation and socio-economic issues. Blackwell Sciences Ltd., Oxford.

Kelleher, K. 2004. Discards in the world's marine fisheries: An update. FAO, Rome.

Leslie, H. M. 2005. A synthesis of marine conservation planning approaches. Conservation Biology **19**: 1701-1703.

Loverich, G. F. 1995. Thinking beyond the traditional codends. Pages 101-106. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Lozano-Montes, H. M., T. J. Pitcher, and N. Haggen. 2008. Shifting environmental and cognitive baselines in the upper Gulf of California. Frontiers in Ecology and Environment **6**: 75-80.

Mackinson, S., and L. Nøttestad. 1998. Combining local and scientific knowledge. Reviews in Fish Biology and Fisheries **8**: 481-490.

Meltzer, L., and J. O. Chang. 2006. Export market influences on the development of the Pacific shrimp fishery of Sonora, Mexico. Ocean and Coastal Management **49**: 222-235.

Neis, B., D. C. Schneider, L. Felt, R. L. Haedrich, J. Fischer, and J. A. Hutchings. 1999. Fisheries assessment: what can be learned from interviewing resource users? Canadian Journal of Fisheries and Aquatic Sciences **56**: 1949-1963.

Perez-Mellado, J. L., and L. T. Findley. 1985. Evaluacion de la Ictiofauna del camarón capturado en las costas de Sonora y norte de Sinaloa, México. Pages 201-254 in A. Yañez-Arancibia, editor. Recursos Pesqueros Potenciales de México: La pesca acompañante del camarón. Universidad Nacional Autonoma, México.

Richardson, E. A., M. J. Kaiser, and G. Edward-Jones. 2005. Variation in fishers' attitudes within an inshore fishery: implications for management. Environmental Conservation **32**: 213-225.

Robins, J. B., M. J. Campbell, and J. G. McGilvray. 1999. Reducing prawn-trawl bycatch in Australia: An overveiw and an example from Queensland. Marine Fisheries Review **61**: 46-55.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, and M. Cariño-Olvera. 2005a. Using fishers' anecdotes, naturalists' observations and grey literature to assess marine species at risk: the case of the Gulf grouper in the Gulf of California, Mexico. Fish and Fisheries **6**: 121-133.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and R. Enríquez-Andrade. 2005b. Rapidly shifting environmental baselines among fishers of the Gulf of California. Proceedings of the Royal Society of London B **272**: 1957-1962.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and J. P. Hawkins. 2006. The value of evidence about past abundance: marine fauna of the Gulf of California through the eyes of 16th to 19th century travellers. Fish and Fisheries **7**: 128-146.

SAGARPA. 2000. Carta Nacional Pesquera.

SAGARPA 2004. Anuario estadístico de pesca 2003. Secretaría de Agricultura, Granaderia, Desarrollo Rural, Pesca y Alimentación. Comisión Nacional de Acuacultura y Pesca., México.

SAGARPA 2006. Anuario Estadístico de Acuacultura y Pesca, Mexico.

Scholz, A., K. Bonzon, R. Fujita, N. Benjamin, N. Woodling, P. Black, and C. Steinback. 2004. Participatory socioeconomic analysis: drawing on fishermen's knowledge for marine protected area planning in California. Marine Policy **28**: 335-349.

Stobutzki, I., S. Blaber, D. Brewer, G. Fry, D. Heales, M. Miller, D. Milton, J. Salini, T. Van der Velde, T. Wassenberg, P. Jones, Y.-G. Wang, M. Dredge, T. Courtenay, K. Chilcott, and S. Eayrs 2000. Ecological sustainability of bycatch and biodiversity in prawn trawl fisheries. FRDC Project 96/257 Final report.

Sumaila, U. R., L. Teh, R. Watson, P. Tyedmers, and D. Pauly. 2008. Fuel price increase, subsidies, overcapacity, and resource sustainability. ICES Journal of Marine Science **65**: 832-840.

Thrush, S. F., J. E. Hewitt, V. J. Cummings, P. K. Dayton, M. Cryer, S. J. Turner, G. A. Funnell, R. G. Budd, C. J. Milburn, and M. R. Wilkinson. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecological Applications **8**: 866-879.

Vendeville, P. 1990. Tropical shrimp fisheries: Types of fishing gear used and their selectivity. FAO, Rome.

Walker, B. 1960. The distribution and affinities of the marine fish fauna of the Gulf of California. Systematic Zoology **9**: 123-133.

Young, R. H., and J. M. Romero. 1979. Variability in the yield and composition of by-catch recovered from Gulf of California shrimping vessels. Tropical Science **21**: 249-264.

Zainudin, I. M., L. Pet-Soede, C. Hitipeuw, and W. Adnyana. 2007. Interactions with sea turtles in Indonesian fisheries - preliminary findings. Indonesian Ocean Turtle Newsletter **6**: 1-10.

5. Clear advice out of great uncertainty – assessing and addressing bycatch of small fishes with limited data [4]

[4] A version of this chapter will be submitted for publication. Foster, S.J. and A.C.J. Vincent. Clear advice out of great uncertainty – assessing and addressing bycatch of small fishes with limited data.

Introduction

Tropical shrimp trawlers accidentally catch and subsequently waste many million tonnes of small fishes each year (Alverson et al. 1994; Kelleher 2004). In some fisheries the small fish are turned into feed for aquaculture and agriculture operations (Clucas 1997; Naylor et al. 2000), whereas in other fisheries the vast majority is discarded back to sea with a very low chance of survival (e.g. Hill & Wassenberg 1990). The waste of such fish needs to be evaluated for sustainability, both because some small fish species are important to their ecosystems, and because they have value as human food, particularly as we fish out larger species (Pauly et al. 1998).

We need to find pragmatic ways to assess and address the impact of nonselective fishing practices on small fishes, especially given our limited knowledge of their life history, population dynamics and ecology (e.g. Stobutzki et al. 2001). A lack of long-term monitoring makes most traditional fisheries methods of assessment intractable. Instead, matrix models may prove useful for assessing a population's status (Morris & Doak 2002). To date, matrix models for marine taxa have been mostly used for charismatic species and/or species of conservation concern (e.g. Crouse et al. 1987; Cortés 1997; Ratner et al. 1997; Cortés 2002; Lewison & Crowder 2003; Wielgus et al. 2007; Curtis & Vincent 2008), as well as species of commercial importance (e.g. case studies in Akçakaya et al. 2004; Diamond et al. 2000). Many of those models were, however, stochastic, requiring information on the variability in life history parameters over time.

With limited life history information, deterministic matrix models can provide a quantitative analysis of a population's status and/or potential efficacy of management options (Morris & Doak 2002). Deterministic matrix models allow a preliminary assessment of a population's anticipated growth or decline under current conditions (Fiedler & Kareiva 1998; Morris & Doak 2002). They can also be used to assess the extent of anthropogenic mortality that is compatible with a population's persistence (Morris & Doak 2002). One of the most useful elements of matrix modelling is elasticity analysis, which evaluates the proportional change in the population growth rate for a proportional change in a vital rate (i.e. survival, growth or reproduction), and thus helps focus future efforts in conservation and research (e.g. Crouse et al. 1987; Crowder et al. 1994; Diamond et al. 2000).

Through the manipulation of vital rates, matrix models may also be used to assess the conservation potential of bycatch mitigation measures that variably affect the vital rates of different size/age classes (Morris & Doak 2002). Fisheries managers can help ensure sustainable exploitation of bycatch species by reducing overall effort, or by reducing bycatch per unit effort with potentially less impact on the fishery (Hall 1995). The latter requires increasing the fisheries selectivity, either through gear specifications or decreeing the timing and location of the fishery. Other potential tools include bycatch or discard quotas, and incentive-based programs, such as increasing total catch limits for fishers with low bycatch to target catch ratios (Horsten & Kirkegaard 2002).

As is true of all models, matrix model outcomes must be considered in light of data limitations and uncertainty. When aspects of a species' life history are unknown or data are uncertain then assumptions are valid, as long as uncertainties in life history parameters are fully explored (Beissinger & Westphal 1998). One benefit to matrix modelling is that the influence of input parameters on model outcomes can be tested through the building of alternative model structures (Akçakaya et al. 2004). Researchers typically test model sensitivities to assumptions and uncertainties in parameters related to survival and reproduction, but in most cases, the influence of other model parameters commonly goes untested (e.g. Naujokaitis-Lewis et al. 2008), and model structures are not usually justified. Studies are still needed to define the limits to the reliability of matrix models in the face of data limitations (Fiedler & Kareiva 1998). Fortunately, however, matrix models do not need perfect accuracy to provide meaningful guidelines for conservation and management (Fiedler & Kareiva 1998). In many cases robust decisions have emerged from such models despite uncertainties (Akçakaya et al. 2004).

The objectives of this study were to use matrix models to determine the population status of a small fish species, silver stardrum (*Stellifer illecebrosus*, Sciaenidae), which is obtained as bycatch in a tropical shrimp trawl fishery in the Gulf of California (Mexico), and then to explore the potential of bycatch mitigation tools to address any impact. Previous work had suggested potential for impact on silver stardrum from its incidental capture in the shrimp trawls (Chapters 2 and 3), and thus we wished to further explore the population status of this species. Data limitations meant that we used deterministic matrix models instead of traditional fisheries models, while recognising that our method, too, has its challenges. Specifically, we set out to do the following: 1) build a deterministic matrix model to estimate the rate of population growth using the best available demographic parameters for the species; 2) conduct elasticity analyses to see which vital rates have greatest relative effects on population growth rate; 3) test the sensitivity of model outcomes to uncertainties in input parameters and model structure, specifically total mortality, growth parameters and the number of age classes modelled; and 4) based on the results of the elasticity analysis, explore potential mitigation tools.

The Gulf of California's industrial shrimp trawl fishery is a model system for studying the impacts of indiscriminate fishing practices on small fish species. The fishery is responsible for the incidental capture of many hundreds of species, the majority of which are small fishes (Perez-Mellado & Findley 1985). In spite of potential for great concern, few published quantitative assessments consider the impact of the shrimp fishery on a species or ecosystem level (Morales-Zárate et al. 2004). The inferred decrease in diversity and biomass of bycatch species over the last 50 years was deduced from limited scientific and anecdotal information (Brusca et al. 2005; Sáenz-Arroyo et al. 2005; Sáenz-Arroyo et al. 2006; Lozano-Montes et al. 2008).

Methods

Study species

The species studied for this research was silver stardrum (Figure 5.1). This species inhabits sandy and muddy bottoms along the Eastern Pacific Ocean from Baja California, Mexico, to Peru (Chao 1995). Our study focused on a population of silver stardrum from the southern Gulf of California, Mexico, where they are commonly obtained as bycatch by industrial shrimp trawlers (Chapter 2).

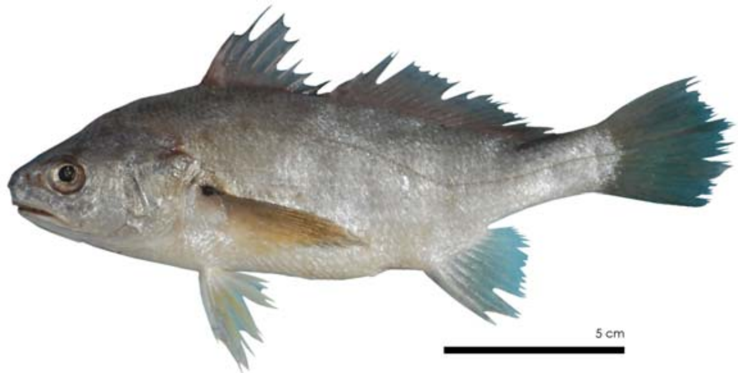

Figure 5.1. Silver stardrum from the southern Gulf of California, Mexico. © CICIMAR.

The vital rates of this species were generated through six months of intensive fisheries dependent sampling on board industrial shrimp trawlers in the southern Gulf of California from September 2006 to March 2007 (see Chapter 2 for details). Most silver stardrum captured during the trawl season (September to March) were immature, but females with ripe eggs were also found throughout this period (Chapter 2).

Matrix model structure and assumptions

To estimate the population's growth rate we used a deterministic density-independent age-structured model (Leslie matrix), where population growth is modelled as a function of age-specific birth rates (fecundities, F_a) and age-specific survival rates (S_a) (Table 5.1). The population growth rate, λ, is the proportional increase in total abundance under stable distribution, no density dependence, no stochasticity and no dispersal (Akçakaya 2005). A population is considered stable when $\lambda = 1$. We only modelled females as the sex ratio is equal for our population (Chapter 2), and there is no evidence to suggest that dynamics differ between males and females. The matrix analysis and all model simulations were carried out using software provided in the popbio package within the statistical program R (R Development Core Team 2004, Stubben & Milligan, 2007, pseudo-code in Appendix II).

Table 5.1. Example Leslie matrix for a population with eight age classes (ages 0 – 7+ years). Age-specific fecundities (F_a) are across the top row of the matrix, and age-specific survival rates (S_a) along the matrix sub-diagonal.

F_0	F_1	F_2	F_3	F_4	F_5	F_6	F_{7+}
S_0	0	0	0	0	0	0	0
0	S_1	0	0	0	0	0	0
0	0	S_2	0	0	0	0	0
0	0	0	S_3	0	0	0	0
0	0	0	0	S_4	0	0	0
0	0	0	0	0	S_5	0	0
0	0	0	0	0	0	S_6	S_{7+}

Life history rates

Growth

We used our existing estimates of the von Bertalanffy Growth Function (VBGF) parameters for silver stardrum to determine age from length for all fish in our sample (N = 5634). Estimates of the VBGF parameters L_{inf} (asymptotic length, 278.05 mm), k (the rate at which the curve approaches L_{inf}, 0.12 per year), and t_0 (the theoretical 'age' of the fish at length zero, -3.50 years) were obtained by fitting the VBGF to data on length at age, where individual age was determined through otolith analysis (Appendix I). We then derived the number and mean total length (L_T, mm) of females at each age (Table 5.2). Three early life history stages (eggs, larvae, and early juvenile fish) were collapsed into one time step, age 0 years. Similarly, small sample sizes led us to group fish seven years of age and older into one class (7+ years). This resulted in a matrix model with eight age classes (ages 0 – 7+ years) (as per Table 5.1).

Table 5.2. Number of females for eight age classes of silver stardrum, with corresponding mean size (total length, L_T, mm and standard deviation, SD), gonado-somatic index (I_G), relative fecundity (I_G of each age class relative to that of age 2), proportion mature, age-specific fecundities (F_a) and survival rates (S_a). 95% confidence interval for S_a: 0.52-0.73.

age (years)	N females	mean L_T	SD L_T	I_G	relative fecundity	Proportion mature	F_a	S_a
0	1414	90.51	11.48	0.000	0.00	0.00	0.00	0.62
1	643	113.20	4.89	0.000	0.00	0.00	0.00	0.62
2	259	131.11	5.00	0.011	1.00	0.30	0.54	0.62
3	143	149.64	4.26	0.014	1.27	0.69	1.60	0.62
4	162	164.01	3.96	0.016	1.49	0.89	2.44	0.62
5	110	176.82	3.32	0.018	1.71	0.96	3.02	0.62
6	60	187.33	2.96	0.020	1.90	0.99	3.44	0.62
7+	27	208.30	19.26	0.025	2.31	1.00	4.24	0.62

Survival

Survival rates were calculated from the instantaneous rate of total mortality (Z) using $S = \exp^{(-Z)}$ (S_a, Table 5.2). We used a linearised length converted catch curve analysis to estimate the instantaneous rate of total mortality (Z, year^{-1}) for our population (Sparre & Venema 1998), as no mark-recapture or long-term data were available to estimate survival rates for this species. The natural logarithms of numbers at age were regressed against age for all fully recruited age classes, and the negative of the resulting slope was assumed equal to Z ($Z = 0.48$ year^{-1}, 95% confidence interval: 0.32-0.65; $F = 56.55$, $DF = 6$, $P < 0.0001$). By modelling Z using a catch curve analysis we assumed a closed population (no emigration/immigration), age constant mortality, age constant catchability (i.e. the sample population is assumed to be a true representation of the actual population), invariance of growth parameters across cohorts (Sparre & Venema 1998; Ernst & Valero 2005), and a stable age distribution. We included the first seven age classes (ages 0 – 6 years) in our analysis as previous research suggested that fish of all ages were retained by the trawl gear (L_T at 50% gear retention = 79.7 mm, thus age at 50% gear retention = 0 years under the above VBGF parameters) (Chapter 2). The last age class was omitted from the analysis because the relationship between length and age suffers greater inaccuracy for large fish (Sparre & Venema 1998).

In absence of a direct estimate, we calculated natural mortality (M, year^{-1}) using the Pauly equation, which derives M from VBGF parameters and the mean annual water temperature where the population resides (Pauly 1980). The mean water temperature across the fishing season during which we collected our samples of silver stardrum was 24.64 °C (S.J.F., unpublished data). Finally, we calculated fishing mortality (F, $year^{-1}$) from Z and M (F = Z – M).

Fecundities

Fecundities were derived from previous knowledge on size at maturity, relative fecundity and proportion mature at size for this species (Chapter 2). Silver stardrum matures at 141 mm, which equates to 2.5 years under the above VBGF parameters, so all individuals aged two years and older were assumed mature. We also assumed individuals in the first age class to be recruits. Silver stardrum females had greater gonado-somatic indices (I_G) as they increased in size (Chapter 2). We took this, and the known proportion mature at size, into account in our calculations of age specific fecundities (F_a, Table 5.2). We assumed a pre-breeding census model, such that fecundities accounted for the survival of recruits to the first age class (S_0) (Akçakaya 2005). Lack of available information meant we had to assume no inter-annual variation in egg survival.

Elasticity analysis

We performed elasticity analyses to determine which vital rate changes would have the greatest effects on λ. We used elasticities instead of sensitivities as elasticities, which tell us the proportional change in λ that would result from a given change in the matrix element, are generally considered more useful for management considerations (Mills & Lindberg 2002). The vital rate in the mean matrix with the highest elasticity is recommended for highest management or research priority (e.g. Crouse et al. 1987; Heppell et al. 1997; Heppell et al. 2000).

Maximum sustainable mortality

We wanted to estimate the value of Z at which λ = 1, the population’s maximum sustainable mortality. To do this we calculated λ 1000 times, each time varying the value of Z from 0.01 to 5 (corresponding to the full range of possible survival rates, 0.01 to 0.99).

Varying the model

We used sensitivity analyses to explore two uncertainties in input parameters and model structure and their effects on λ: L_{inf} and the number of age classes modelled (N_A).

Although our model was based on the best available information, we were concerned about the uncertainty of several model parameters and assumptions. First, we were uncertain about our value for L_{inf} as sampled silver stardrum attained sizes larger than the predicted value of L_{inf} (maximum L_T in sample = 305 mm, predicted L_{inf} = 278 mm). Second, our decision to combine all individuals aged seven years and older into one age class was somewhat arbitrary. Varying L_{inf} and N_A would change both the number and average size of fish in a given age class, which in turn would affect both survival rates (which depend on the numbers at age) and fecundities (which depend on the numbers and average size of individual at age).

We evaluated the influence of model uncertainties on the estimated population trajectory by conducting the matrix analysis 1000 times, each time with a varied value for L_{inf} and N_A. We varied L_{inf} from 220 to 350 mm, corresponding to the largest individual in our aged sample to the upper 95% confidence interval of the fitted growth model (Appendix I). We varied N_A from 3 to 11; when N_A equalled 3 all mature individuals were assumed to contribute equally to recruitment, and setting the upper limit to 11 ensured a sufficient sample size within the last age class. For each value of L_{inf} the parameters k and t_0 were estimated by fitting the VBGF (with fixed L_{inf}) to length at age data using a non-linear search function (Crawley 2007). Using the estimates for L_{inf}, k and t_0, we then re-calculated age from length for all the fish in our sample, grouping all individuals older than N_A into one final age class. We calculated survival rates and fecundities as above, populated the Leslie matrix, ran the matrix analysis, and exported the matrix elements, λ, elasticities, M and F to a spreadsheet for later analysis. We examined the influence of L_{inf} and N_A on model predictions in two separate sensitivity analyses: once where Z was modelled with a catch curve analysis, and again when it was assumed known (and so held constant at 0.48 $year^{-1}$, see ***Survival*** above). Finally, we repeated our analysis of maximum sustainable mortality, this time assuming the VBGF estimates to be correct, but varying N_A.

The influence of L_{inf}, N_A and Z on λ was evaluated with a Spearman Rank Correlation. Ranked correlation coefficients can be used as measures of relative influence of input parameters on response variables (e.g. Curtis & Naujokaitis-Lewis 2008). All statistical analyses were carried out using Prism 5, Version 5.2 (GraphPad Software Inc.).

Mitigating impact

Elasticity analysis for all model variations revealed that juvenile survival rates (S_0 and S_1) had the most influence on λ for our population of silver stardrum (see **Results**, below). We thus wanted to determine the expected magnitude of increase in λ for a given increase in juvenile survival rates, and how the relationship would change with different assumptions about N_A.

For this analysis we fixed the VBGF parameters (L_{inf}, k, t_0) to the best estimates (see ***Growth***, above, and Appendix I), and then repeated the matrix analysis 1000 times, each time varying N_A (from 3-11) and thus Z, as it was derived with a catch curve analysis. With each simulation, we also increased the values of S_0 and S_1 by between 5 and 70% over adult survival (S_a). Linear regression analyses were used to test the significance of the relationship between λ and % increase in S_a.

Results

Matrix model

When the best available growth parameters for silver stardrum were used to populate the Leslie matrix, and the matrix structure was assumed to be eight age classes, the population was estimated to be capable of increasing in size (λ = 1.11). F was estimated at 0.26 year^{-1}, slightly higher than the estimate for M (0.22 year^{-1}).

Elasticity analysis

Elasticity values suggested that survival rates had the greatest influence on λ, particularly survival of juveniles – age 0 and 1 year individuals (S_0 and S_1, Figure 5.2b). The elasticity values of survival rates were large compared to values for fecundities, but among fecundities those of age 3 year individuals (F_3) had the greatest influence on λ (Figure 5.2a).

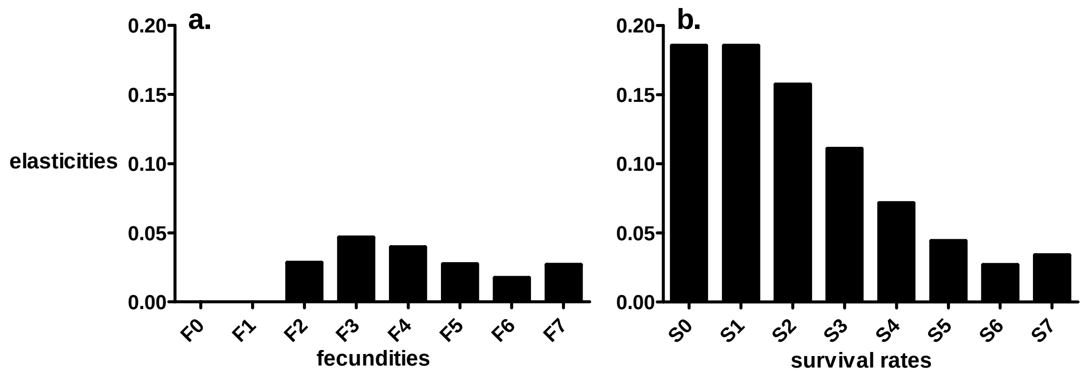

Figure 5.2. Deterministic elasticity of population growth rate (λ) to changes in values of vital rates, a) fecundities, and b) survivals, used in a matrix model for silver stardrum.

Maximum sustainable mortality

The maximum sustainable total mortality (Z at which λ approximated 1) was estimated at 0.61 year^{-1}, a 27% increase over estimated Z (0.48 year^{-1}), but within the 95% confidence intervals (see ***Survival***, above).

Varying the model

Deriving Z from a catch curve analysis

Varying L_{inf} and N_A, while deriving Z (and so survival rates) using a catch curve analysis, had a clear impact on our estimates of λ for this species (Figure 5.3a, Table 5.3). Variations in λ were mainly driven by Z, which in turn was mainly determined by the number of age classes used in the analysis (N_A) (Figure 5.4). Input values for L_{inf} had much less of an influence on λ (Table 5.3). When the N_A classes was varied, and Z was derived from a catch curve analysis, bounded estimates for Z were wide, even for a single set of VBGF parameters. Z ranged from 0.43 – 0.98 year^{-1}, with a mean value of 0.62 ± 0.12 year^{-1} across all L_{inf} and N_A combinations.

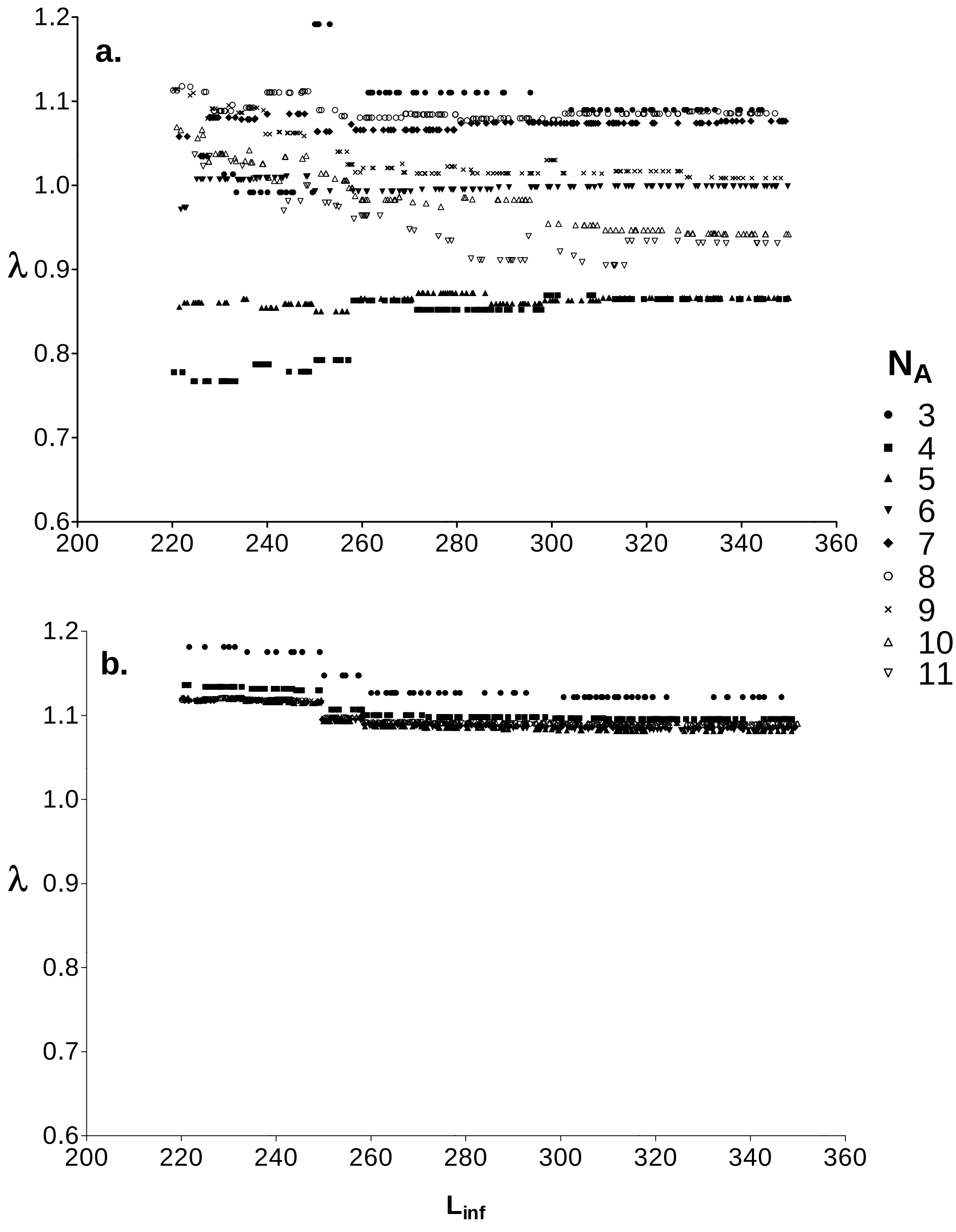

Figure 5.3. Estimates of population growth rate (λ) of silver stardrum across a range of values for asymptotic length (L_{inf}, mm), and number of age classes used in the analysis (N_A). In a) total mortality was estimated using a catch curve analysis, and thus was highly dependent on N_A, and in b) total mortality was held constant, and so independent of N_A.

Table 5.3. The relative influence of input parameters on the population growth rate of silver stardrum. Z = total mortality ($year^{-1}$), N_A = number of age classes included in the analysis, L_{inf} = asymptotic length (mm), and CL = confidence limit.

Parameter	Number of pairs	Spearman r	Lower 95% CL	Upper 95% CL	P value
Z	1000	-0.96	-0.96	-0.95	< 0.0001
N_A	1000	0.28	0.22	0.33	< 0.0001
L_{inf}	1000	-0.05	-0.11	0.02	0.1382

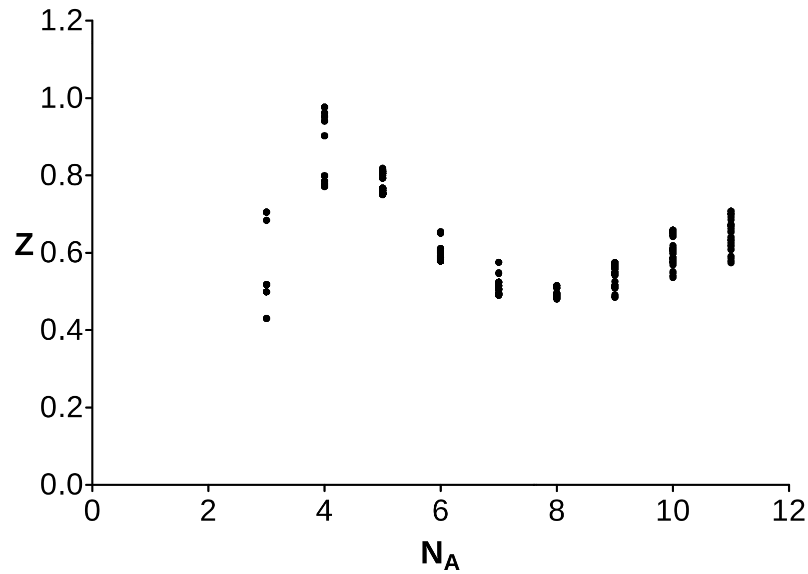

Figure 5.4. Estimated total mortality (Z, $year^{-1}$) for a population of silver stardrum where a variable number of age classes, N_A, were included in the catch curve analysis used to estimate Z. Variations at a given N_A are the result of varying the growth parameters for this species.

Population growth rate (λ) ranged from 0.77 – 1.19 across all L_{inf} and N_A combinations, and from 0.84 to 1.08 for a given N_A (and so averaged across L_{inf} values). Averaging across L_{inf} values, the highest estimate for λ occurred when $N_A = 3$, the scenario in which all mature individuals were assumed to contribute equally to recruitment. In addition, when $N_A = 3$, L_{inf} had the greatest influence on λ (Figure 5.3a).

Estimates for M only depended on L_{inf}, and not N_A. M decreased with increasing L_{inf}, and ranged from 0.16 to 0.34 year^{-1} across all L_{inf} values (Figure 5.5). As a result, estimated F increased as L_{inf} increased, although variations in Z resulting from varying N_A somewhat obscured this pattern (Figure 5.6a).

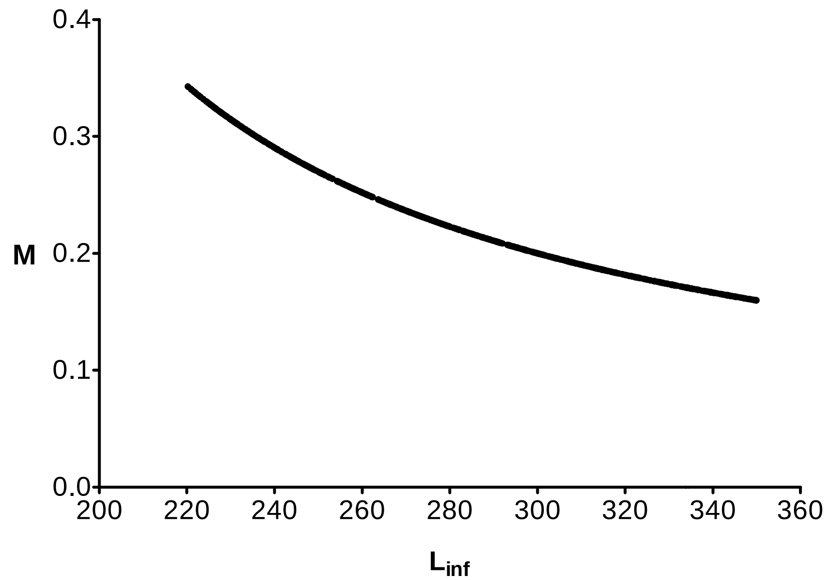

Figure 5.5. Estimated natural mortality (M, year^{-1}) for a population of silver stardrum across a range of values for asymptotic length (L_{inf}, mm, a parameter of The von Bertalanffy Growth Function for fishes). M was calculated using the Pauly equation (Pauly 1980).

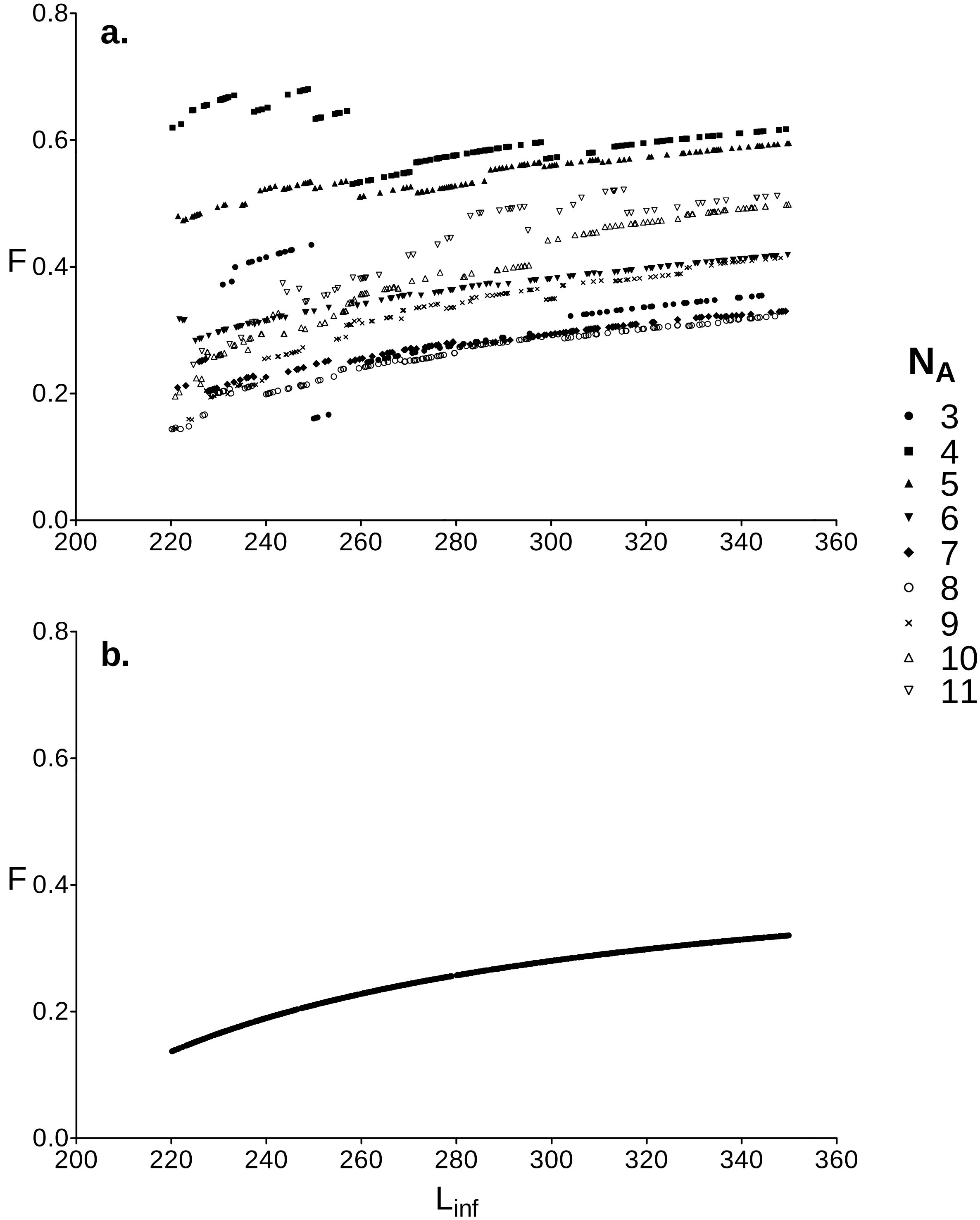

Figure 5.6. Estimated fishing mortality (F, year^{-1}) of a population of silver stardrum across a range of values for asymptotic length (L_{inf}, mm). In a) F was deduced from a situation where total mortality (Z) was varied by changing the number of age classes, N_A, used in the catch curve analyses, and in b) F was deduced from a situation where Z was assumed known, and fixed at 0.48 year^{-1}. F was derived from Z and natural mortality, which was calculated using the Pauly equation (Pauly 1980).

When Z is assumed known

The population growth rate (λ) of silver stardrum did not vary nearly as much when we fixed the value of Z instead of deriving it from a catch curve analysis, but in this case L_{inf} had more influence on λ than did N_A (Figure 5.3b, Table 5.4). When Z was assumed known, estimated values of λ ranged from 1.08 to 1.18 across all L_{inf} and N_A combinations, and from 1.09 to 1.14 for a given N_A when averaged across L_{inf} values. Population growth rate (λ) was still greatest, though also most variable across L_{inf} values, when $N_A = 3$. There appeared to be a threshold value for L_{inf}, of about 250 mm, above and below which the mean predicted values of λ were different (Figure 5.3b). This resulted from differing predictions for the number of recruits – the estimated number of recruits for simulations with $L_{inf} < 250$ mm was 2828, and for $L_{inf} > 250$ mm was 2560 – and would, in turn, have affected estimated fecundities.

Estimates for M were the same as in the previous analysis, where Z varied, as they only depended on L_{inf}, and not N_A (Figure 5.5). But because Z was held constant, the increase in F with increasing L_{inf} values was much clearer under these circumstances (Figure 5.6b).

Table 5.4. The relative influence of input parameters on the population growth rate of silver stardrum. L_{inf} = asymptotic length (mm), N_A = number of age classes included in the analysis, and CL = confidence limit.

Parameter	Number of Pairs	Spearman r	Lower 95% CL	Upper 95% CL	P value
L_{inf}	1000	-0.64	-0.68	-0.60	< 0.0001
N_A	1000	-0.16	-0.22	-0.10	< 0.0001

Elasticity analysis

All model variations were consistent in revealing that juvenile survival rates had the most influence on λ for our population of silver stardrum. The mean elasticity values associated with the matrix elements across all L_{inf} and N_A combinations revealed that S_0 and S_1 were the most important determinants of λ, followed by a steady decline in elasticity values for survival rates as age class progressed (Figure 5.7). The pattern was the same whether Z was varied (determined from a catch curve analysis, Figure 5.7.I) or assumed known (held constant, Figure 5.7.II). There were only very small differences in elasticity values between the scenarios.

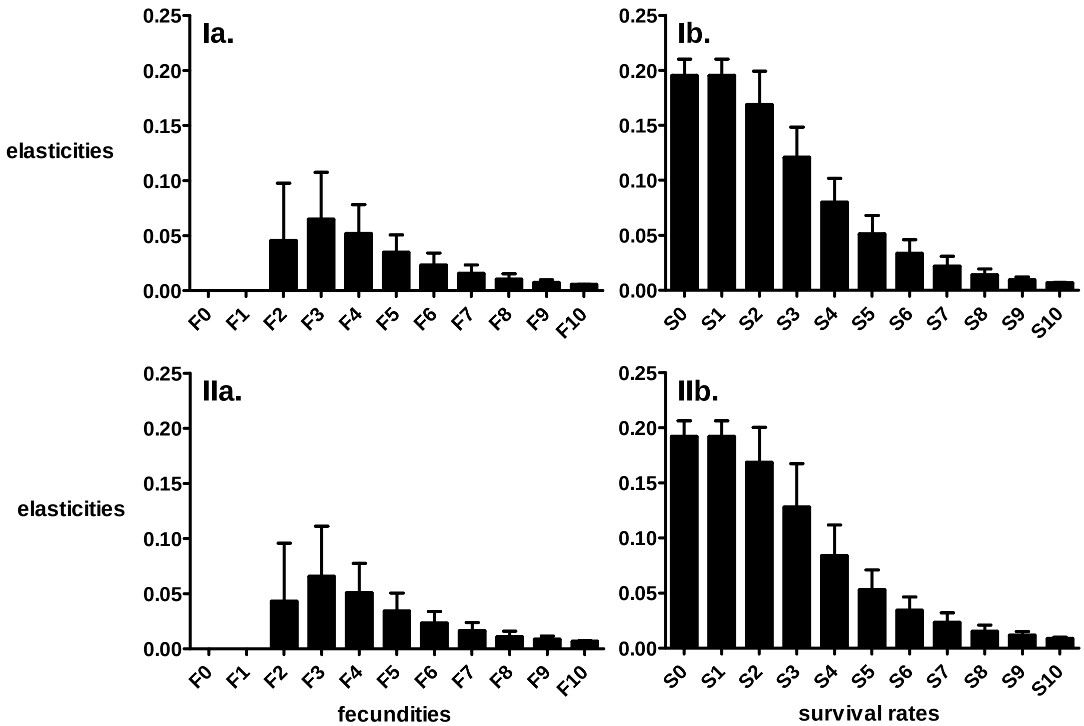

Figure 5.7. Deterministic elasticity of population growth rate (λ) to changes in values of vital rates, a) fecundities and b) survivals, used in a matrix model for silver stardrum where total mortality was I) varied, and II) held constant.

Maximum sustainable mortality

Varying N_A, but holding the VBGF parameters constant, had only a small impact on the estimated maximum sustainable mortality for our population. The value of Z at which λ approximated 1 was more or less constant across assumed number of age classes, with a mean value of 0.59 ± 0.02 $year^{-1}$.

Mitigating impact

There was a significant relationship between an increase in juvenile survival (measured as % increase in S_a) and λ for all values of N_A (Table 5.5). Our analyses also revealed that while the slope of the relationship differed significantly among our assumptions for N_A (and thus Z; F = 30665.9, DF = 982, P < 0.0001), but only the analysis where N_A = 3 had a slope very different from the other simulations (Table 5.5).

Table 5.5. The relationship between increased juvenile survival rate (relative to adult survival) and population growth rate for silver stardrum. N_A = number of age classes included in the analysis and Z = total mortality ($year^{-1}$).

N_A	Z	slope	standard deviation	R^2	DF	F	P value
3	0.79	0.0064	7.8370E-06	1.00	65	673420	< 0.0001
4	0.85	0.0031	5.0610E-06	1.00	140	378709	< 0.0001
5	0.79	0.0031	4.7350E-06	1.00	131	423211	< 0.0001
6	0.58	0.0035	4.8910E-06	1.00	131	520101	< 0.0001
7	0.50	0.0037	5.2500E-06	1.00	119	509086	< 0.0001
8	0.48	0.0038	5.0400E-06	1.00	114	561328	< 0.0001
9	0.56	0.0035	4.9480E-06	1.00	111	512833	< 0.0001
10	0.60	0.0034	4.9900E-06	1.00	121	476100	< 0.0001
11	0.66	0.0033	7.2260E-06	1.00	50	209495	< 0.0001

The average slope across simulations where N_A equalled 4 to 11 was 0.0034. This result can be used to explore potential management measures for our population. For example, our earlier analysis varying L_{inf}, N_A and Z predicted a range of λ values from 0.77 – 1.19. If we were to proceed with extreme caution, we would assume that the lowest value for λ (0.77) is the true value. Halting the decrease of our population would therefore call for an increase in juvenile survival of 81% over the adult rate. Such an increase in S_0 and S_1 would theoretically increase λ to a value greater than 1 (1.05).

Discussion

Despite great uncertainty regarding the impact of industrial shrimp trawling on silver stardrum in the southern Gulf of California, our study indicates that any precautionary management should focus on increasing the survival of younger age classes. This result adds support to claims that deterministic matrix models may be useful for conservation and management where parameters can only be estimated crudely (Fiedler & Kareiva 1998; Akçakaya et al. 2004), as in the case of small non-targeted fish species. Nonetheless, our study also supports the case for acquiring robust estimates of total mortality, and is a rare demonstration of the importance of understanding age-based changes in vital rates to the predictions of matrix models.

Our study confirms the importance of testing the effect of uncertainties in mortality estimates on model outcomes. Our highly variable estimates of the population trajectory for silver stardrum, from declining to increasing in size, arose primarily because of such uncertainties with respect to total mortality (and hence survival rates). Our study thus confirms the risk of using catch curve analyses to estimate total mortality where there are doubts about meeting all model assumptions, as would be the case for most natural systems (Ernst & Valero 2005). Indeed, we cannot be sure to have met any of the assumptions in the current study. Ideally one would use data obtained over a number of years, or from a tag-recapture program (Robson & Chapman 1961). In the absence of these types of data, fisheries assessment manuals suggest obtaining an initial estimate of mortality through a length converted catch curve analysis on samples of a population's length frequency (e.g. Sparre & Venema 1998; Cochrane 2002; King 2007). Using catch curve analyses to estimate mortality may, however, err on the side of conservation; in cases where assumptions were violated, catch curve analyses tended to overestimate mortality rates (Ernst & Valero 2005).

Given the great uncertainty in our mortality rates, we should be cautious when interpreting our population's estimated maximum sustainable mortality. The maximum sustainable mortality for our population of silver stardrum was estimated to be higher than our first estimate for total mortality, but fell within the confidence interval, and was similar in magnitude to the mean of mortality estimates across all model simulations. The current vague estimates of total mortality suggest, therefore, that any increase in fishing pressure on this species would be unwise until a more robust estimate can be obtained. At the same time, our elasticity analysis confirmed the importance of obtaining reliable mortality estimates.

The elasticity analyses across all model simulations were consistent in suggesting that survival rates, especially those of juveniles, were the most important determinants of the population's trajectory. Unfortunately, obtaining sound estimates of juvenile survival is one of the most challenging tasks in fisheries science (Morgan 2007). Estimating juvenile survival is frequently time and cost intensive, and thus is not likely to be made a priority for small fishes of no commercial value. That notwithstanding, the elasticity analysis provided clear guidance that reducing the mortality of young fish would be the most precautionary management measure with respect to silver stardrum.

A variety of management measures might lead to reduced fishing mortality on smaller/younger individuals. The obvious approach of increasing the mesh size used for the trawl nets is sound in theory, but unlikely to work in practice. First, existing regulations in the world's tropical shrimp trawl fisheries, including that of the Gulf of California, are commonly ignored as targeted shrimps have become smaller over time (Gillett 2008; Davies et al. 2009; Chapter 4). Second, the meshes of trawl nets frequently become clogged with debris, such that even the smallest of individuals can not escape (S.J.F., pers. obs.). Instead, gear closures in areas of vulnerable life history stages may be the most pragmatic way forward. Spatio-temporal closures could be considered where species demonstrate an ontogenetic distribution in time and/or space, as is the case with many fishes (Field et al. 2006), including our population of silver stardrum (Chapter 3). Gear closures have the added benefit of having the support of the majority of the Gulf's industrial trawl fishers whom we consulted about potential solutions for the problems facing their fishery (Chapter 4).

In addition to improving estimates of mortality, our study also suggests the importance of understanding age-specific variations in vital rates. It is notable that simulations where we (unrealistically) assumed that all mature individuals contributed equally to recruitment, using only three age classes, consistently and unreliably predicted the highest rates of population growth. This is cause for concern where information on age specific variations in survival and fecundities are unknown, as in the case of a matrix model for the European mudminnow (*Umbra krameri*) (Wanzenböck 2004). The researchers knew that fecundity changed with female size, but did not want to sacrifice the endangered species to obtain age specific data on fecundity. They were thus forced to use a model without stage structure, and may well have overestimated population growth for this species.

Except for the case of using only three age classes, varying the number of age classes of mature individuals did not greatly affect the predicted population growth rate. Such a result might not hold for other species or populations, though, so it is important to test the effects of different model structure with regard to age classes and/or to justify choices in that respect. In theory, age classes should be collapsed if all grouped individuals behave similarly with respect to vital rates (survival and fecundities) (Beissinger & Westphal 1998). Usually, however, collapses are done due to sample size limitations (e.g. Smedbol & Stephenson 2004) or without any stated justification (e.g. Fu et al. 2004; Hart & Cadrin 2004; Nicol & Todd 2004)

As is often the case, our model demonstrates that estimates of VBFG parameters (here L_{inf}) can heavily affect the perceived relative importance of fishing mortality (F) on a population. For the matrix model, varying L_{inf} had the greatest impact on model outcomes where it changed the observed proportion at age and thus the estimated number of recruits, and thus fecundities. However, where L_{inf} mattered most was not the matrix model, but in our estimation of natural mortality (M) using the Pauly equation (Pauly 1980). Natural mortality estimates ranged widely over plausible values for L_{inf}, and thus so did the perceived relative importance of F for our population. Many fisheries analyses require an estimate for M, but obtaining direct estimates is difficult where populations have already been fished. Researchers must instead rely on methods that estimate M indirectly from growth characteristics of the species (e.g. Pauly 1980; Hoenig 1983), recognising that these methods "rank no higher than qualified guesses" (Sparre & Venema 1998). Inaccurate estimates of L_{inf} can arise where the aged sample does not cover the entire size range of the actual population (as was the case for this study), where sampling is not truly representative, and/or where there are ageing errors. As well as influencing F, L_{inf} is sometimes used to derive other life history parameters in absence of direct estimates (e.g. Froese & Binohlan 2000, 2003), so should be deduced carefully.

Although our sensitivity analyses showed that increasing juvenile survival should increase population growth rate, other factors come into play in bottom trawling situations. The assumptions we make in our study are justified – given that our intention was to estimate λ and its sensitivity to vital rates, and not to predict future population abundance – but were almost certainly violated, at least to some extent (Beissinger & Westphal 1998). This is important to consider because even when on average population growth exceeds 1, a population can be highly vulnerable if growth rates vary in time (Morris & Doak 2002). Variations in vital rates over time for bycatch species may be occurring if, in addition to fishing mortality, they are suffering degradation and/or loss of their habitats from bottom trawling. Increasing juvenile survival may not achieve increases in population growth rates in situations where critical habitat is limited (Green and Hirons 1991 as cited in Fiedler & Kareiva 1998). Variations may also occur where a decrease in trawl effort results in an increase in predatory fish, and therefore in predation on smaller fish species. Thus, the effects of any trawl closures on small fish species and their con-specifics should be monitored to allow for adaptive management.

Literature cited

Akçakaya, H. R. 2005. RAMAS GIS: Linking spatial data with population viability analysis (version 5). Applied Biomathematics, Setauket, New York.

Akçakaya, H. R., M. A. Burgman, O. Kindvall, C. C. Wood, P. Sjögren-Gulve, J. S. Hatfield, and M. A. McCarthy, editors. 2004. Species Conservation and Management: Case Studies. Oxford University Press, New York.

Alverson, D. L., M. H. Freeberg, J. G. Pope, and S. A. Murawski. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper No. 339: 1-233.

Beissinger, S. R., and M. I. Westphal. 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management **62**: 821-841.

Brusca, R. C., L. T. Findley, P. A. Hastings, M. E. Hendrickx, J. T. Cosio, and A. M. van der Heiden. 2005. Macrofaunal diversity in the Gulf of California. Pages 179-202 in J.-L. E. Cartron, G. Ceballos, and R. S. Felger, editors. Biodiversity, Ecosystems, and Conservation in Northern Mexico. Oxford University Press, Inc., New York, New York.

Chao, L. N. 1995. Sciaenidae. Corvinas, barbiches, bombaches, corvinatas, corvinetas, corvinillas, lambes, pescadillas, roncachos, verrugatos. Pages 1427-1518 in W. Fischer, F. Krupp, W. Schneider, C. Sommer, K. E. Carpenter, and V. Niem, editors. Guía FAO para Identificación de Especies para lo Fines de la Pesca. Pacifico Centro-Oriental. FAO, Rome.

Clucas, I. 1997. A study of the options for the utilization of bycatch and discards from marine capture fisheries. Page 59. FAO Fisheries Circular No. 928. Food and Agriculture Organisation of the United Nations, Rome.

Cochrane, K. L., editor. 2002. A fishery manager's guidebook - Management measures and their application. Food and Agriculture Organisation of the United Nations (FAO), Rome.

Cortés, E. 1997. A stochastic stage-based population model of the sandbar shark in the Western North Atlantic. Pages 115-136 in J. A. Musick, editor. Life in the Slow Lane: Ecology and Conservation of Long-Lived Marine Animals. American Fisheries Society Symposium 23, Methesda, Maryland.

Cortés, E. 2002. Incorporating uncertainty into demographic modeling: Application to shark populations and their conservation. Conservation Biology **16**: 1048-1062.

Crawley, M. J. 2007. The R Book. John Wiley & Sons Ltd., Chichester, UK.

Crouse, D. T., L. B. Crowder, and H. Caswell. 1987. A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology **68**: 1412-1423.

Crowder, L. B., D. T. Crouse, S. Heppell, and T. H. Martin. 1994. Predicting the impact of turtle excluder devices on Loggerhead sea turtle populations. Ecological Applications **4**: 437-445.

Curtis, J. M. R., and I. Naujokaitis-Lewis. 2008. Sensitivity of population viability to spatial and nonspatial parameters using GRIP. Ecological Applications **18**: 1002-1013.

Curtis, J. M. R., and A. C. J. Vincent. 2008. Use of population viability analysis to evaluate CITES trade-management options for threatened marine fishes. Conservation Biology **22**: 1225-1232.

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

Diamond, S. L., L. B. Cowell, and L. B. Crowder. 2000. Population effects of shrimp trawl bycatch on Atlantic croaker. Canadian Journal of Fisheries and Aquatic Sciences **57**: 2010-2021.

Ernst, B., and J. L. Valero. 2005. Bias of equilibrium-based estimators under biological and fishery disequilibria. Pages 715-735 in G. H. Kruse, V. F. Gallucci, D. E. Hay, R. I. Perry, R. M. Peterman, T. C. Shirley, P. D. Spencer, B. Wilson, and D. Woodby, editors. Fisheries Assessment and Management in Data-Limited Situations. Alaska Sea Grant College Program.

Fiedler, P. L., and P. M. Kareiva, editors. 1998. Conservation Biology for the Coming Decade. Chapman & Hall, London.

Field, J. C., A. E. Punt, R. D. Methot, and C. J. Thomson. 2006. Does MPA mean 'Major Problem for Assessments'? Considering the consequences of place-based management systems. Fish and Fisheries **7**: 284-302.

Froese, R., and C. Binohlan. 2000. Empirical relationships to estimate asymptotic length, length at first maturity and length at maximum yield per recruit in fishes, with a simple method to evaluate length frequency data. Journal of Fish Biology **56**: 758-773.

Froese, R., and C. Binohlan. 2003. Simple methods to obtain preliminary growth estimates for fishes. Journal of Applied Ichthyology **19**: 376-379.

Fu, C., C. C. Wood, and J. Schweigert. 2004. Pacific Herring (*Clupea pallasi*) in Canada. Pages 256-270 in H. R. Akçakaya, M. A. Burgman, O. Kindvall, C. C. Wood, P. Sjögren-Gulve, J. S. Hatfield, and M. A. McCarthy, editors. Species Conservation and Management: Case Studies. Oxford University Press, New York.

Gillett, R. 2008. Global study of shrimp fisheries. Food and Agriculture Organization of the United Nations, Rome.

Hall, M. A. 1995. Strategic issues in managing bycatch. Pages 29-39. Solving bycatch: Considerations for today and tomorrow: proceedings of the Solving Bycatch Workshop. University of Alaska Sea Grant College Program, Seattle, WA.

Hart, D. R., and S. X. Cadrin. 2004. Yellowtail Flounder (*Limanda ferruginea*) off the northeastern United States. Pages 230-244 in H. R. Akçakaya, M. A. Burgman, O. Kindvall, C. C. Wood, P. Sjögren-Gulve, J. S. Hatfield, and M. A. McCarthy, editors. Species Conservation and Management: Case Studies. Oxford University Press, New York.

Heppell, S., H. Caswell, and L. B. Crowder. 2000. Life histories and elasticity patterns: Perturbation analysis for species with minimal demographic data. Ecology **81**: 654-665.

Heppell, S., L. B. Crowder, and T. R. Menzel. 1997. Life table analysis of long-lived marine species with implications for conservation and management. Pages 115-136 in J. A. Musick, editor. Life in the Slow Lane: Ecology and Conservation of Long-Lived Marine Animals. American Fisheries Society Symposium 23, Methesda, Maryland.

Hill, B. J., and I. R. Wassenberg. 1990. Fate of discards from prawn trawlers in Torres Straight. Australian Journal of Marine and Freshwater Research **41**: 53-64.

Hoenig, J. M. 1983. Empirical use of longevity data to estimate mortality rates. Fishery Bulletin U.S. **82**: 898–903.

Horsten, M. B., and E. Kirkegaard. 2002. Bycatch from a perspective of sustainable use. Page 16. IUCN SSC European Sustainable Use Specialist Group, Brussels.

Kelleher, K. 2004. Discards in the world's marine fisheries: An update. FAO, Rome.

King, M. 2007. Fisheries Biology, Assessment and Management. Blackwell Publishing, Oxford, UK.

Lewison, R. L., and L. B. Crowder. 2003. Estimating fishery bycatch and effects on a vulnerable seabird population. Ecological Applications **13**: 743-753.

Lozano-Montes, H. M., T. J. Pitcher, and N. Haggen. 2008. Shifting environmental and cognitive baselines in the upper Gulf of California. Frontiers in Ecology and Environment **6**: 75-80.

Mills, L. S., and M. S. Lindberg. 2002. Sensitivity analysis to evaluate conservation actions. Pages 338-366 in S. R. Beissinger, and D. R. McCullough, editors. Population Viability Analysis. The University of Chicago Press, Chicago.

Morales-Zárate, M. V., F. Arreguín-Sánchez, J. Martínez-López, and S. E. Lluch-Cota. 2004. Ecosystem trophic structure and energy flux in the Northern Gulf of California, México. Ecological Modelling **174**: 331-345.

Morgan, S. K. 2007. The ontogenetic ecology and conservation of exploited tropical seahorses. PhD. McGill University, Montreal.

Morris, W. F., and D. F. Doak 2002. Quantitative Conservation Biology - Theory and Practive of Population Viability Analysis. Sinauer Associates, Inc., Sunderland, Massachusetts, USA.

Naujokaitis-Lewis, I. R., J. M. R. Curtis, P. Arcese, and J. Rosenfield. 2008. Sensitivity analyses of spatial population viability analysis models for species at risk and habitat conservation planning. Conservation Biology **23**: 225-229.

Naylor, R. L., R. J. Goldburg, J. H. Primavera, N. Kautsky, M. C. M. Beveridge, J. Clay, C. Folke, J. Lubchenco, H. Mooney, and M. Troell. 2000. Effect of aquaculture on world fish supplies. Nature **405**: 1017-1024.

Nicol, S., and C. Todd. 2004. Trout Cod (*Muccullochella macquariensis*) in the Murray River, Southereast Australia: Prioritizing efforts. Pages 219-229 in H. R. Akçakaya, M. A. Burgman, O. Kindvall, C. C. Wood, P. Sjögren-Gulve, J. S. Hatfield, and M. A. McCarthy, editors. Species Conservation and Management: Case Studies. Oxford University Press, New York.

Pauly, D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES Journal of Marine Science **39**: 175-192.

Pauly, D., J. Dalsgaard, V. Christensen, R. Froese, and F. J. Torres. 1998. Fishing down marine food webs. Science **279**: 860-863.

Perez-Mellado, J. L., and L. T. Findley. 1985. Evaluacion de la Ictiofauna del camarón capturado en las costas de Sonora y norte de Sinaloa, México. Pages 201-254 in A. Yañez-Arancibia, editor. Recursos Pesqueros Potenciales de México: La pesca acompañante del camarón. Universidad Nacional Autonoma, México.

R Development Core Team. 2004. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ratner, S., R. Lande, and B. Roper. 1997. Population viability analysis of spring chinook salmon in the South Umpqua River, Oregon. Conservation Biology **11**: 879-889.

Robson, D. S., and D. G. Chapman. 1961. Catch curves and mortality rates. Transactions of the American Fisheries Society **90**: 181-189.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and R. Enríquez-Andrade. 2005. Rapidly shifting environmental baselines among fishers of the Gulf of California. Proceedings of the Royal Society of London B **272**: 1957-1962.

Sáenz-Arroyo, A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and J. P. Hawkins. 2006. The value of evidence about past abundance: marine fauna of the Gulf of California through the eyes of 16th to 19th century travelers. Fish and Fisheries **7**: 128-146.

Smedbol, R. K., and R. L. Stephenson. 2004. Atlantic Herring (*Clupea harengus*) in the Northwest Atlantic Ocean. Pages 245-255 in H. R. Akçakaya, M. A. Burgman, O. Kindvall, C. C. Wood, P. Sjögren-Gulve, J. S. Hatfield, and M. A. McCarthy, editors. Species Conservation and Management: Case Studies. Oxford University Press, New York.

Sparre, P., and S. Venema. 1998. Introduction to tropical fish stock assessment - Part I: Manual. FAO Fisheries Technical Paper. FAO, Rome. www.fao.org/docrep/W5449E/w5449e00.htm.

Stubben, C. & Milligan, B. (2007). Estimating and analyzing demographic models using the popbio package in R. *Journal of Statistical Software* **22**.

Wanzenböck, J. 2004. European Mudminnow (*Umbra krameri*) in the Austrian floodplain of the River Danube. Pages 200-207 in H. R. Akçakaya, M. A. Burgman, O. Kindvall, C. C. Wood, P. Sjögren-Gulve, J. S. Hatfield, and M. A. McCarthy, editors. Species Conservation and Management: Case Studies. Oxford University Press, New York.

Wielgus, J., F. Ballentyne IV, E. Sala, and L. R. Gerber. 2007. Viability analysis of reef fish populations based on limited demographic information. Conservation Biology **21**: 447-454.

6. General discussion

This first explicit attempt to assess the impacts of shrimp trawling on small fishes in bycatch has revealed the real challenges of such an endeavour, to the point where full understanding is probably unobtainable. Six months of intensive sampling followed by many more months of data management and analysis produced only limited information on just a few of the hundreds of species obtained as bycatch in one tropical shrimp trawl fishery. It will be close to impossible to repeat my analyses for the hundreds of small fish species obtained as bycatch in this particular fishery, let alone for the thousands captured by tropical shrimp trawl fisheries worldwide. Moreover, even were such information obtained, it would only allow for suggestions of potential fishing effects, with tremendous additional effort required to explore any causal relationship between shrimp trawling and the population status of these species.

Despite the many difficulties, my thesis has produced original information on understudied species, information which indicates that at least some small fishes may indeed be vulnerable to trawl pressures. My research therefore argues and substantiates the need to implement trawl free areas as pragmatic and precautionary methods for reducing potential impacts on small fishes.

Assessing impact

I achieved the first objective of my thesis, to use newly derived life history information for several small fish species to evaluate possible effects of their incidental capture, although I could not deduce effects with any certainty. My application of length based indicators (Chapter 2) and qualitative criteria (Chapters 2 and 3) suggest that that some small fishes show potential for overfishing. We thus need to re-evaluate the status of small species, which have generally been considered resilient to fishing pressures (e.g. Jennings et al. 1998; Jennings et al. 1999a; Jennings et al. 1999b); in contrast, research has traditionally focused on larger species with lower productivity. Comparing among my case study taxa indicates that small fish species obtained as bycatch in the Gulf of California shrimp trawl fishery varied in their life history and population characteristics, and so probably differ in their vulnerabilities to fishing pressure (Chapters 2 and 3). The same will be true across many small species caught in many shrimp trawl fisheries.

My research showed that appropriate life history information, where it can be deduced, can provide insight into fisheries impacts (Chapters 2, 3 and 5). The information I obtained on size and reproduction was particularly useful for investigating potential for fishing effects on my taxa (Chapters 2 and 3). Comparing sizes of fish retained by the gear to length at maturity, an important length-based reference point, raised concern for bigscale goatfish and silver stardrum, as most of these fish in bycatch were apparently immature (Chapter 2). On the other hand, the majority of sandperch, and almost all lumptail searobin, were caught above length at maturity (Chapter 2). Comparing numbers and sizes over time suggested that fishing may be exerting a lower toll on bigscale goatfish than on silver stardrum: juveniles of the former recruited to the study area throughout the fishing season despite the ongoing trawl fishery while the occurrence and density of the latter declined across the season (Chapter 3). My research thus identified one species, silver stardrum, for which its incidental capture by the Gulf's industrial shrimp trawlers could be a problem, although further studies would be needed to clarify and confirm impact.

Despite the utility of life history data, the majority of my analyses, and thus any inferences about impact of shrimp trawling, were limited by a dearth of such information for my species. Pragmatism forced me to limit my original research ambitions to six of the hundreds of small fish species obtained as bycatch in the Gulf's fishery but even so I only produced novel information for four species, and a detailed analysis of just one. The life history information I managed to derive was limited by the fisheries dependent and short-term ('snap-shot') nature of my study. In particular, I could not confirm my samples to be truly representative of their populations for any of my species (Chapters 2, 3 and 5), and could not obtain a robust estimate of mortality for silver stardrum (Chapter 5). These shortcomings limited the use of demographic modelling to assess the population status of silver stardrum (Chapter 5). In addition, sampling only half the year gave incomplete pictures of growth and reproductive cycles (Chapter 2), and even taxonomic confusions hindered my research (Chapter 2).

The data limitations I faced are quite common in fisheries, even where all available resources are employed. Current fisheries stock assessment methods are data hungry and require extensive expertise (e.g. Hilborn & Walters 1991; Walters & Martell 2004), and are thus inaccessible to most of the world's fisheries (Mahon 1997; Berkes 2003). Indeed, we lack the necessary data for these models – including long-term data on catches, abundance, and size – for even for the most valuable of target species (Mace 2004). Consequently, there is a call for new fisheries management techniques that require few data but are precautionary and robust (e.g. Mahon 1997; Johannes 1998; FAO 2008). Several of the methods I explored, including length based indicators, qualitative criteria, and demographic modelling approaches, are hailed as alternative assessment methods for data-limited situations (e.g. Trenkel & Rochet 2003; Dulvy et al. 2004; Froese 2004). However, the effort required to obtain these data makes it unlikely that even these 'simple' approaches will be employed for any but our most valued stocks or species of conservation concern. Even then, confirming fisheries effects with any certainty would require a time series of population information, fisheries independent sampling, and/or the ability to overlay spatio-temporal distributions of fishing effort on my distribution data (e.g. Andrew & Pepperell 1992).

Socio-economic realities

I achieved the second objective of my thesis, to shed light on the social dimensions of tropical shrimp fisheries management. My novel examination of shrimp trawl fishers' perceptions and attitudes found that, among the problems facing the Gulf of California's shrimp trawl fishery, fishers tended to distance themselves from responsibility for management options by identifying problems generated externally (Chapter 4). Any solutions to the Gulf's industrial shrimp fishery are definitely going to depend on proper enforcement and reliable governance, issues generally overlooked in existing literature addressing its management (e.g. Meltzer & Chang 2006; Gillett 2008). Certainly, none of the biological information will make any difference to fisheries management unless there is also political and social will for change (e.g. Ludwig et al. 1993; Browman et al. 2004; Hilborn 2004).

Should the situation in the Gulf remain *status quo*, the majority of fishers I interviewed considered that the fishery has no future (Chapter 4). This is probably true of many of the worlds tropical shrimp trawl fisheries, given that most are economically overfished, and require considerable help from their governments to persist (Gillett 2008). While an end to shrimp trawling in the Gulf, and elsewhere, might benefit the small fishes for which there is no targeted fishery, it would also displace effort onto other marine resources (Chapter 4). Research is needed to understand what the potential end to shrimp trawl fisheries means for the environment and the people that currently depend on them. Further, research is needed to increase our understanding of motivations for subsidies to shrimp trawl fisheries, and how such funds could be better used to improve the situation of the environment and its stakeholders – such as increasing alternative employment opportunities, or enforcing existing fisheries regulations.

The majority of fishers I interviewed supported trawl closures as a way to alleviate problems associated with the fishery, as long as effective enforcement were put in place (Chapter 4). Such enforcement would require a top-down commitment from fisheries regulatory agencies (Chapter 4). It should, however, also be supported by involving fishers in site selection, and the implementation of self-policing through co-management (Defeo & Pérez-Castañeda 2003). My contribution of fishers' perceptions of important fishing grounds should provide a useful starting point for increasing the social acceptance of any closures (Chapter 4). Since I only consulted a small fraction of the Gulf's fishers, it would be wise to repeat my consultations more broadly to ensure the views held by study respondents are consistent with those of the larger fisher population.

Addressing impact

I achieved the third objective of my thesis, to consider the potential value of socially acceptable bycatch mitigation measures for small fishes. My research suggests that trawl free areas are both biologically appropriate and socially acceptable measures for alleviating problems associated with non-selective fishing of small fishes (Chapters 3, 4 and 5). The great diversity of small species in bycatch and their size overlap with targeted shrimps limits the potential to increase selectivity through technological changes (Brewer et al. 1998; Cochrane 2002, Chapters 2 and 3), and means that management methods designed to address one species might not work for another (Winemiller & Rose 1992).

My research suggests that a number of trawl restrictions covering a range of depths and latitudes would be required to mitigate the fishery's impact on small fish species (Chapters 3). The two species I analysed – a small fraction of the many obtained as bycatch in the Gulf's shrimp trawls – predictably exhibited distribution patterns that were at odds with one another. As a result I was not able to single out a particular range of latitudes or depths to recommend for protection. The two species did share a common trend that spatial variables mattered more than temporal ones, suggesting that in addition to being diverse in space, trawl closures would need to be permanent (as apposed to seasonal).

My matrix model, while of limited use for assessing impact, provided support for trawl closures to mitigate potential effects of shrimp trawling on silver stardrum (Chapter 5). Despite great uncertainty regarding the population status of silver stardrum, my elasticity analyses indicated that any precautionary management should focus on increasing the survival of younger age classes. Although a variety of management measures might help (e.g. increasing mesh sizes), reduced fishing mortality on smaller/younger individuals would best be achieved by trawl closures in habitats supporting smaller individuals. When we consider socioeconomic impacts along with ecological goals, my finding suggests that technological changes are unlikely to achieve comparable levels of protection (Chapter 4). Given the diversity in ontogenetic distribution patterns of my case study species (Chapter 3), and after accounting for all small fish species, we would again need a wide array of closures covering the complete range of exploited depths.

Pragmatic and precautionary methods of reducing fisheries impacts are going to be useful in a wide array of fisheries management, not just with reference to the incidental capture of small fishes. Protected areas are recommended in other data-limited situations such as small scale fisheries (Mahon 1997), and have been used by tropical fishing cultures for years without formal data (Johannes 1998). Most importantly, protected areas should help bet hedge against uncertainty, especially for fisheries that negatively affect habitats and non-target species (Botsford et al. 1997; Lauck et al. 1998; Defeo & Pérez-Castañeda 2003; Hilborn et al. 2004). Closing areas to fishing when the effects of extractive activities are uncertain would be appropriately precautionary, giving priority to conservation of resource without which there will be no social or economic benefits (Mace 2001).

Trawl closures in the Gulf of California may be broadly useful. To date gear closures have been implemented as fisheries management tools (Cochrane 2002), while complete area closures have been promoted for ecosystem conservation (Agardy 2000; Mosquera et al. 2000; Mace 2001). However, many of the reasons for implementing gear closures (Cochrane 2002) are consistent with the objectives for ecosystem-based management (Gislason et al. 2000). In addition to reducing bycatch, trawl closures may help to address several of the direct and indirect issues associated with tropical shrimp fisheries. It is well established that industrial fisheries (and many small-scale) are overcapitalised, with too many fishers chasing too few fish (Mahon 1997; Mace 2001; Hilborn et al. 2003), and tropical shrimp trawl fisheries are no exception (Gillett 2008). Trawl closures thus have the potential to improve the fisheries profitability, which would be a valuable incentive for their implementation. To date, however, relatively few studies have examined the effectiveness of partial-take marine protected areas (but see Murawski et al. 2000; Ye et al. 2000; Denny & Babcock 2004; McClanahan et al. 2006). Research is needed to understand the benefits of trawl closures for both fisheries and ecosystem conservation, thereby supporting adaptive management of these fisheries.

An emerging concern

Even more worrying than the waste of small fish species in bycatch might be their incautious use. The vast majority of the small fish caught as bycatch by the Gulf of California's shrimp trawlers are discarded at present (Gillett 2008). We need to avoid a situation where the Gulf's shrimp fishery becomes an unmanaged but targeted multi-species "trash" fishery, to meet the demands for feed for Mexico's growing aquaculture industry. Such a shift would reduce incentives for mitigating the fishery's environmental and social impacts. Indeed, this is a global issue. As the catch of commercial shrimp species declines, trawl owners are likely to turn to the high-volume and low-profit "trash" industry to stay afloat (Lobo et al. 2009). Several questions related to these fisheries need immediate attention:

- We need to understand better the fate of retained bycatch and its economic role. While we know that 36% of targeted small fishes are turned into fishmeal and fishoil for use in aquaculture and agriculture (Alder et al. 2008), little is known about the small fish species caught incidentally but retained for the same uses.

- We need to increase our understanding of the impacts of a transition from unprofitable shrimp trawl fisheries to multi-species trawl fisheries that target small fishes. These fisheries continue to be labelled as shrimp fisheries, but shrimp are only a small fraction of the total catch. Retaining the small fish (so reducing discards) does not reduce the population and ecological impacts.

- Research is needed to evaluate the lost opportunities for human food supply where small fishes are turned into fishmeal and fishoil for aquaculture and agriculture operations, instead of being fed directly to people.

Conclusions

Although I have shown the difficulty in determining the exact nature of the threat of trawling to individual small fish species using fisheries dependent sampling and a snap-shot of life history parameters, the intensity of trawl effort and the wide variety of life histories among such species mean that some (at least) will experience a detrimental impact. As just one example, I had hoped to include seahorses as a study taxon but their populations had been so depleted, apparently by shrimp trawling (Baum & Vincent 2005, Chapter 4), that too few samples were available. Given the long history of the Gulf of California shrimp trawling and its exhaustive effort, populations of many other small species may also have been very diminished well before my study began. Ironically, the analytical approaches I took here required me to select species that offered significant sample sizes and thus might have been most robust to considerable extraction. Even so, I was limited in what I could deduce or infer. For example, although my study raises concern for populations of bigscale goatfish and silver stardrum that experience high incidental capture, their relatively high abundance compared to other bycaught species may suggest they are resilient to fishing pressure.

In closing, I emphasise that analytical challenges cannot be allowed to justify inaction in resource management, whether for small fish species in bycatch or in any other context. If we want, we can, to some extent, use a combination of biological prediction (Dulvy et al. 2004) and stakeholder perception (Chapter 4) to identify species that need focused technical research on population status, and then apply some of the approaches I used here to develop and evaluate management tools (Chapters 2, 3 and 5). In reality, however, my research has shown that such detailed work will simply not be feasible for the vast majority of small fish species, and that managers will do far better to take broadly useful and acceptable steps, such as area or temporal closures. Such closures, where implemented in a framework of adaptive management, will allow for the impacts of trawl fishing on small fishes to be further deduced, while also bet-hedging against uncertainty.

Literature cited

Agardy, T. 2000. Effects of fisheries on marine ecosystems: a conservationist's perspective. ICES Journal of Marine Science **57**: 761-765.

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

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

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

Berkes, F. 2003. Alternatives to conventional management: lessons from small-scale fisheries. Environments **31**: 1-19.

Botsford, L. W., J. C. Castilla, and C. H. Peterson. 1997. The management of fisheries and marine ecosystems. Science **277**: 509-515.

Brewer, D., N. Rawlinson, S. Eayrs, and C. Burridge. 1998. An assessment of bycatch reduction devices in a tropical Australian prawn trawl fishery. Fisheries Research **36**: 195-215.

Browman, H. I., P. M. Cury, R. Hilborn, S. Jennings, H. K. Lotze, P. M. Mace, S. A. Murawski, D. Pauly, M. Sissenwine, K. I. Stergiou, and D. Zeller. 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Marine Ecology Progress Series **274**: 269-303.

Cochrane, K. L., editor. 2002. A fishery manager's guidebook - Management measures and their application. Food and Agriculture Organisation of the United Nations (FAO), Rome.

Defeo, O., and R. Pérez-Castañeda. 2003. Misuse of marine protected areas for management: The case of Mexico. Fisheries **28**: 35-36.

Denny, C. M., and R. C. Babcock. 2004. Do partial marine reserves protect reef fish assemblages. Biological Conservation **116**: 119-129.

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

FAO. 2008. Toward integrated assessment and advice in small-scale fisheries. FAO Fisheries and Aquaculture Technical Paper **515**: 1-84.

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

Gillett, R. 2008. Global study of shrimp fisheries. Food and Agriculture Organization of the United Nations, Rome.

Gislason, H., M. Sinclair, K. Sainsbury, and R. O'Boyle. 2000. Symposium overview: incorporating ecosystem objectives within fisheries management. ICES Journal of Marine Science **57**: 468-475.

Hilborn, R. 2004. Ecosystem-based fisheries management: the carrot or the stick? Marine Ecology Progress Series **274**: 275-278.

Hilborn, R., T. A. Branch, B. Ernst, A. Magnusson, C. V. Minte-Vera, M. D. Scheuerell, and J. L. Valero. 2003. State of the world's fisheries. Annual Review of Environment and Resources **28**: 359-399.

Hilborn, R., K. Stokes, J.-J. Maguire, T. Smith, L. W. Botsford, M. Mangel, J. Orensanz, A. Parma, J. C. Rice, J. Bell, K. L. Cochrane, S. Garcia, S. J. Hall, G. P. Kirkwood, K. Sainsbury, G. Stefansson, and C. Walters. 2004. When can marine reserves improve fisheries management? Ocean and Coastal Management **47**: 197-205.

Hilborn, R., and C. J. Walters 1991. Quantitative Fisheries Stock Assessment: Choices, Dynamics and Uncertainties. Chapman and Hall, New York.

Jennings, S., S. P. R. Greenstreet, and J. D. Reynolds. 1999a. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology **68**: 617-627.

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

Jennings, S., J. D. Reynolds, and N. V. C. Polunin. 1999b. 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.

Lauck, T., C. W. Clark, M. Mangel, and G. R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications **8**: S72-S78.

Lobo, A. S., R. Arthur, A. Manica, and A. Balmford. 2009. Chickenfeed: Fishing for scraps in an unpredictable sea. International Marine Conservation Congress. All Academic, Inc., Fairfax.

Ludwig, D., R. Hilborn, and C. Walter. 1993. Uncertainty, resource exploitation, and conservation: Lessons from history. Science **260**: 17+36.

Mace, P. M. 2001. A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management. Fish and Fisheries **2**: 2-32.

Mace, P. M. 2004. In defense of fisheries scientists, single-species models and other scapegoats: confronting the real problems. Marine Ecology Progress Series **274**: 285-291.

Mahon, R. 1997. Does fisheries science serve the needs of managers of small stocks in developing countries? Canadian Journal of Fisheries and Aquatic Sciences **54**: 2207-2213.

McClanahan, T. R., M. J. Marnane, J. E. Cinner, and W. E. Kiene. 2006. A comparison of marine protected areas and alternative management approaches to coral-reef management. Current Biology **16**.

Meltzer, L., and J. O. Chang. 2006. Export market influences on the development of the Pacific shrimp fishery of Sonora, Mexico. Ocean and Coastal Management **49**: 222-235.

Mosquera, I., M. I. Côte, S. Jennings, and J. D. Reynolds. 2000. Conservation benefits of marine reserve for fish populations. Animal Conservation (The Zoological Society of London) **4**: 321-332.

Murawski, S. A., R. Brown, H. L. Lai, P. J. Rago, and L. Hendrickson. 2000. Large-scale closed areas as a fisheries - management too in temperate marine systems: the Georges bank experience. Bulletin of Marine Science **66**: 775-798.

Trenkel, V. M., and M.-J. Rochet. 2003. Performance of indicators derived from abundance estimates for detecting the impact of fishing on a fish community. Canadian Journal of Fisheries and Aquatic Sciences **60**: 67-85.

Walters, C. J., and S. J. D. Martell 2004. Fisheries Ecology and Management. Princeton University Press, Princeton, NJ.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life-history diversification in North-American Fishes: Implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences **49**: 2196-2218.

Ye, Y., A. H. Alsaffar, and H. M. A. Mohammed. 2000. Bycatch and discards of the Kuwait shrimp fishery. Fisheries Research **45**: 9-19.

7. Appendices

Appendix I: Assessing growth of silver stardrum through otolith analysis

Context

Determining the age of individual silver stardrum (*Stellifer illecebrosus*) in my samples was a necessary step toward furthering the understanding of potential for impact from its incidental capture by the Gulf of California industrial shrimp trawl fishery. Information on silver stardrum life history and population parameters, presented in Chapters 2 and 3, suggested that its incidental capture may have negative consequences for a population of silver stardrum in the southern Gulf of California. I wished to further explore the potential for impact using matrix models (Chapter 5). Matrix models require estimates of a species growth parameters in order to estimate the age schedules of mortality and reproduction, which in turn are used to assess population status. To my knowledge, silver stardrum had never been aged before.

Age of individual silver stardrum was determined through otolith analysis. Otoliths are small calcified structures found in heads of fish used for balance and orientation. Otoliths grow in tandem with the fish, forming yearly rings (annuli) like those of a tree. Counting these rings allowed me to determine age. Otoliths were collected during the 2006-2007 shrimp trawl season. The field research portion of this project was funded by the International Development Research Council of Canada. The EJLB Foundation provided generous funding for the age-analysis portion of my project.

Research

Sample collection

Samples of silver stardrum were collected from the bycatch of two industrial shrimp trawlers, at 38 stations along the coasts of Sinaloa and Nayarit, southern Gulf of California, Mexico. Details of sample collection can be found in Chapter 2.

For the purposes of determining the age of the fish in my samples, I removed the sagittal otoliths from the first (up to) 10 individuals from each of sample tow. The otoliths were cleaned and stored in numbered plastic vials, and then all (up to) 10 vials were placed in a well labelled plastic bag, indicating station and trip. I collected otoliths from 1073 individuals of silver stardrum from September 2006 to March 2007.

Reading the otoliths

I used the "break and burn" method to age individual silver stardrum in my samples. The suitability of this method for ageing this species was confirmed by experts at the Fish Aging Unit, Pacific Biological Station, Nanaimo, British Columbia. The otoliths were processed in a lab at The University of British Columbia.

First, a glycerine/water solution was added to the vials containing the otoliths to increase their transparency and thus the visibility of the annuli. Approximately one week later, the whole otolith was dried and weighed, and then cut in half using a low speed diamond saw. Using forceps, one of the otolith halves was held over the flame of an alcohol burner until it became caramel coloured. This process allowed the protein otolin to react with the heat, and display the annuli. The burnt otolith half was placed flat side up in a dish of clay, and examined under a dissecting microscope. The annuli were counted and an age estimate recorded for the sample being examined.

As no clear criteria existed for ageing silver stardrum using the "break-and-burn" method, I had to develop criteria for interpreting the ring patterns of the otoliths. It was crucial that the criteria were consistently applied to all otolith samples. To this end, at least every fifth otolith was set aside and independently examined by a second age reader. Ages assigned by the first reader were compared with those assigned by the second reader. I recorded the percent agreement between the two readers.

In addition to assigning age, I explored whether there existed a relationship between age and otolith weight, as has been observed in other species (e.g. Choat et al. 2003; Lou et al. 2005). Such a relationship would greatly assist in ageing these species in the future, as only otolith weight would have to be recorded, and then translated into age using the relationship between the two variables.

In summary, a research assistant and I were able to "read" otoliths from 897 individuals of silver stardrum ranging in total length from 44 to 222 mm. Although the otoliths were collected across space and time, for the purposes of this analysis all data were pooled.

Modelling growth

Individual silver stardrum in our samples ranged in age from 0 – 8 years, with the average age of the sampled population estimated at 1.7 ± 1.9 years (Figure I.1).

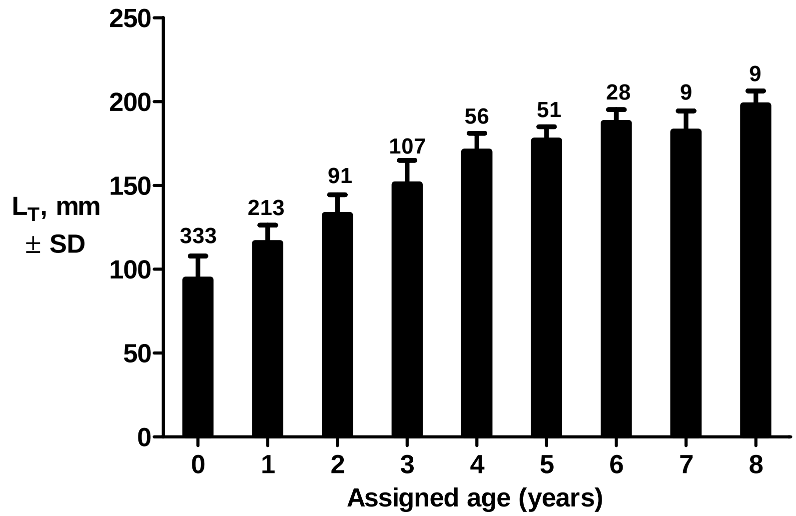

Figure I.1. Number and average size of silver stardrum individuals assigned to each of nine age classes. L_T = total length, SD = standard deviation.

Accuracy of otolith readings for this species decreased with increasing size class to about 150 mm (Figure I.2). This may have occurred because annuli get closer together as a fish grows, making it harder to distinguish among them. Increased accuracy at the largest size classes was probably due to readers taking cues from the large size of the otoliths. We re-read 19.2% of the otoliths for this species, with readers agreeing 69.2% of the time. The average difference in assigned age between readers was 1.1 ± 0.5 years.

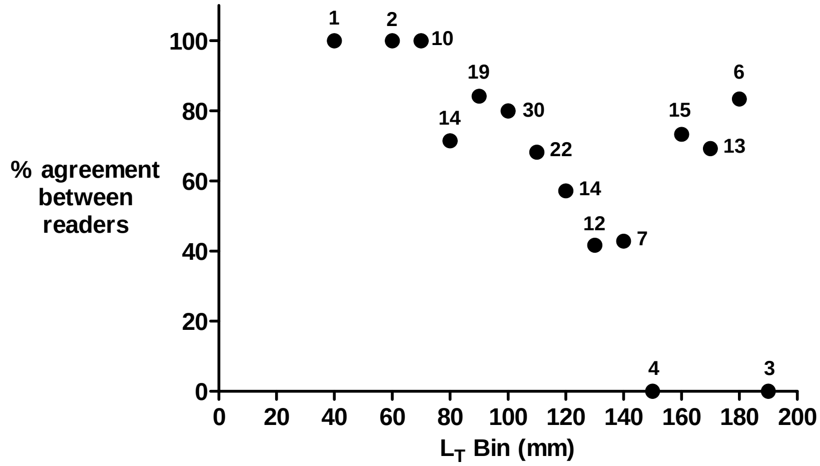

Figure I.2. Percent agreement between readers over size classes for otoliths of silver stardrum. Sample sizes for each size class are shown next to data points. L_T = total length.

Otolith weight can be used as a proxy for age in this species as a clear relationship was found between otolith weight and assigned age ($R^2 = 0.92$, $P < 0.0001$, DF = 923; assigned age = 0.06 * otolith weight - 0.70) (Figure I.3). This result will prove very useful for further studies on this species.

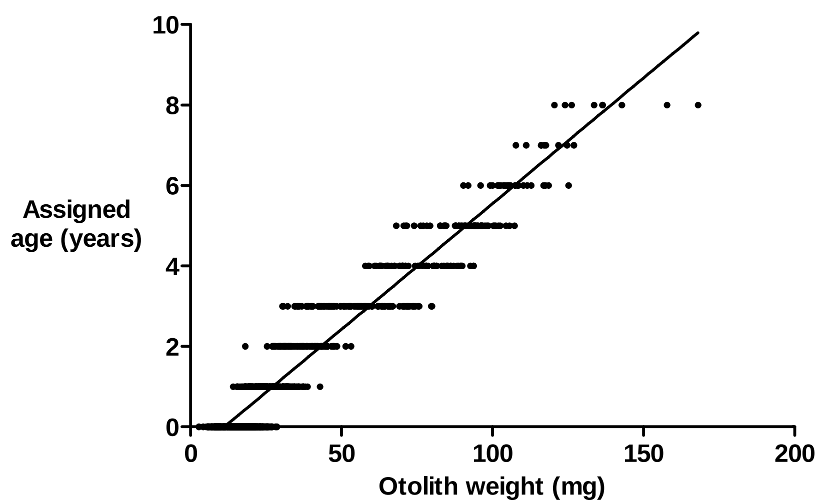

Figure I.3. Assigned age, using the "break and burn" method, versus otolith weight for <u>silver stardrum</u>. $R^2 = 0.92$.

I was able to use the length at age data for silver stardrum to model the average growth rates of individuals in the population. Growth functions predict the length of a fish as a function of its age. The model used a multinomial likelihood to estimate growth, treating size-at-age observations as multinomial samples, with expected catches in each size-age category dependent on growth parameters, growth variation, abundance at age, and population mortality (Taylor et al. 2005). The model estimated the von Bertalanffy Growth Function (VBGF) parameters L_{inf} (asymptotic length), k (the rate at which the curve approaches L_{inf}), t_0 (the theoretical 'age' the fish at length zero), total mortality rate (Z) and the coefficient of variation in length at age (cv) (Table I.1). Likelihood profiling was used to determine the 5 and 95% confidence intervals for each parameter (Hilborn & Mangel 1997) (Table I.1).

Sensitivity analyses for each of the parameters revealed that values for L_{inf}, k and t_0 were sensitive to changes in values of the same parameters (L_{inf}, k and t_0), but insensitive to changes in Z and cv. On the other hand, Z and cv were largely insensitive to changes in other parameter values.

Table I.1. Estimated values for asymptotic length (L_{inf}, mm), the rate at which the curve approaches L_{inf} (k), the theoretical 'age' of fish at length zero (t_0, years), total mortality rate (Z, year^{-1}) and the coefficient of variation in length at age (cv) for a population of silver stardrum from the southern Gulf of California, Mexico. CL = confidence limit.

Parameter	Estimate	lower 95% CL	upper 95% CL	Notes
L_{inf}	278.1	253.0	345.5	profiled from 150 mm to 400 mm at step size of 0.5 (upper bound chosen as felt anything larger to be unrealistic) – upper limit not well defined
k	0.12	0.09	0.15	profiled from 0.05 to 0.55 at step size of 0.005 – lower limit not well defined
t_0	-3.50	-3.95	-3.20	profiled from -0.10 to -5.0 at step size of 0.05 – lower limit not well defined
cv	0.114	0.110	0.118	profiled from 0.07 to 0.14 at step size of 0.001 – well defined
Z	0.42	0.39	0.44	profiled from 0.3 to 0.6 at step size of 0.002 – well defined

Literature cited

Choat, J. H., D. R. Robertson, J. L. Ackerman, and J. M. Posada. 2003. An age-based demographic analysis of the Caribbean stoplight parrotfish *Saprisoma viride*. Marine Ecology Progress Series **246**:265-277.

Hilborn, R., and M. Mangel. 1997. Alternative views of the scientific method and modeling. Pages 12-38. The ecological detective: Confronting models with data. Princeton University Press, Princeton, NJ.

Lou, D. C., B. D. Mapstone, G. R. Russ, C. R. Davies, and G. A. Begg. 2005. Using otolith weight-age relationships to predict age-based metrics of coral reef fish populations at different spatial scales. Fisheries Research **71**:279-294.

Taylor, N. G., C. J. Walters, and S. J. D. Martell. 2005. A new likelihood function for simultaneously estimating von Bertalanffy growth parameters, gear selectivity, and natural and fishing mortality. Canadian Journal of Fisheries and Aquatic Sciences **62**:215-223.

Appendix II: Pseudo-code for matrix model (Chapter 5)

Pseudo-code for R

load data
load R library "popbio"

For 1000 reps:

1. Determine age of fish in sample

Select L_{inf} from range of values (220 – 305 mm)
    L_{inf} = asymptotic length

Estimate of k and t_0 by fitting the VBGF (with fixed L_{inf}) to length at age data using a non-linear search function
    $TL_i = L_{inf} * \{1-\exp[-k*(t_i-t_0)]\}$
    where TL_i = total length at age i, k = curvature parameter, t_i = age i, and t_0 = age of length 0.

Select number of age classes to be used in the analysis (from 3-11)
    N_A = assumed number of age classes

Use VBGF parameter estimates to calculate age from length for all fish in sample
    $i = (\ln[1-(TL/L_{inf})]/-k) + t_0$
    Round i to 0 decimals (so integer)
    Use loop to fix age classes: where TL > L_{inf}, age = N_A; where predicted age > N_A, age = N_A; and where assigned age < 0, age = 0

NOTE: For base model VBGF parameters are fixed at best estimates, and N_A = 7

2. Survival rates

Where total mortality (Z) is determined from catch curve analysis:
    Count number of fish per age class (N_i)
    Calculate $\ln(N_i)$ per age class = $\ln(N_i)$
    Linear regression of $\ln(N_i)$ on age : -(slope) = Z **exclude last age class from regression as contains individuals of more than one age
    Calculate survival: $S = \exp^{(-Z)}$

Where Z is assumed known:
    Z = 0.48

To determine threshold mortality:
    Z = a random amount from 0.01 to 5

To vary survival of juveniles:

Increase S_0 and S_1 by a random amount from 5 – 30% over S

To calculate M and F:

Use VBGF parameters to calculate M based on Pauly equation:

$M = 10^{[-0.0066-(0.279*LOG(L_{inf}))+(0.6543*LOG(k))+(0.463*LOG(T = 24.64))]}$

$F = Z - M$

3. Fecundity

Calculate mean TL per age class: mTL_i

Calculate mean GSI at age:

$G_i = [(mTL_i * 0.0007) + 0.012]^2$

Calculate relative GSI at age:

$RGSI_i = GSI_i/GSI_3$

**Replace $RGSI_0$ and $RGSI_1$ with 0 (as not reproductively mature)

Calculate proportion mature at age:

$P_i = 1/\{1 + [\exp(-0.09) * (mTL_i – 140.81)]\}$

Calculate reproducing females:

Nirepro = Nifemales*Pi

Recruits = Nifemales[1] (number females in first age class)

Calculate fecundity of fish in first mature age class (age = 2 years):

F2 = Recruits/sum(RGSIi*Nirepro)

Calculate rest of fecundities:

Fi = F2 * RGSIi * Pi

4. Matrix analysis

Build Leslie matrix:

F_i across top, S along diagonal (and lower right entry)

Run eigen analysis – output lambda and elasticities to spreadsheet

REPEAT

Appendix III: Behavioural Research Ethics Board approval

The University of British Columbia
Office of Research Services and Administration
Behavioural Research Ethics Board

Certificate of Approval

PRINCIPAL INVESTIGATOR	DEPARTMENT	NUMBER
Vincent, A.C.J.	Fisheries	**B05-1057**

INSTITUTION(S) WHERE RESEARCH WILL BE CARRIED OUT

UBC Campus ,

CO-INVESTIGATORS:

Foster, Sarah, Fisheries

SPONSORING AGENCIES

Chocolaterie Guylian

TITLE :

Impacts of Shrimp Trawl Fisheries on Small Fish Species

APPROVAL DATE	TERM (YEARS)	AMENDMENT:	AMENDMENT APPROVED:
06-01-10 (yr/mo/day)	1	Jan. 12, 2006, Consent form	JAN 23 2006

CERTIFICATION:

The request for continuing review of an amendment to the above-named project has been reviewed and the procedures were found to be acceptable on ethical grounds for research involving human subjects.

Approved on behalf of the Behavioural Research Ethics Board
by one of the following:
Dr. Peter Suedfeld, Chair,
Dr. Susan Rowley, Associate Chair
Dr. Jim Rupert, Associate Chair
Dr. Arminee Kazanjian, Associate Chair

This Certificate of Approval is valid for the above term provided there is no change in the experimental procedures

The University of British Columbia
Office of Research Services and Administration
Behavioural Research Ethics Board

Certificate of Approval

PRINCIPAL INVESTIGATOR	DEPARTMENT	NUMBER
Vincent, A.C.J.	Fisheries	**B05-1057**

INSTITUTION(S) WHERE RESEARCH WILL BE CARRIED OUT
UBC Campus ,
CO-INVESTIGATORS:
Foster, Sarah, Fisheries
SPONSORING AGENCIES
Chocolaterie Guylian
TITLE :
Impacts of Shrimp Trawl Fisheries on Small Fish Species

APPROVAL DATE	TERM (YEARS)	DOCUMENTS INCLUDED IN THIS APPROVAL:
JAN 10 2006	1	Jan. 3, 2006, Contact letter / Nov. 24, 2005, Consent form

CERTIFICATION:

The application for ethical review of the above-named project has been reviewed and the procedures were found to be acceptable on ethical grounds for research involving human subjects.

Approved on behalf of the Behavioural Research Ethics Board
by one of the following:
Dr. Peter Suedfeld, Chair,
Dr. Susan Rowley, Associate Chair
Dr. Jim Rupert, Associate Chair

This Certificate of Approval is valid for the above term provided there is no change in the experimental procedures

Appendix IV: Animal Care Ethics Board approval

The University of British Columbia

Animal Care Certificate

Application Number: A06-0021

Investigator or Course Director: Amanda C.J. Vincent

Department: Fisheries

Animals Approved: Fish

Start Date: **February 1, 2006** Approval Date: **February 1, 2006**

Funding Sources:

Funding Agency:	Chocolaterie Guylian
Funding Title:	Project Seahorse
Funding Agency:	Natural Sciences and Engineering Research Council
Funding Title:	Ecosystem effects of different fisheries restrictions on coral reefs
Unfunded title:	N/A

The Animal Care Committee has examined and approved the use of animals for the above experimental project.

This certificate is valid for one year from the above start or approval date (whichever is later) provided there is no change in the experimental procedures. Annual review is required by the CCAC and some granting agencies.