Bangor University

DOCTOR OF PHILOSOPHY

Hydroacoustics as a tool for the assessment of fishes within existing and candidate marine protected areas (MPAs)

Egerton, Jack

Award date:
2017

Awarding institution:
Bangor University

Link to publication

Hydroacoustics as a tool for the assessment of fishes within existing and candidate marine protected areas (MPAs)

PRIFYSGOL

BANGOR

UNIVERSITY

A thesis presented to the University of Bangor for the degree of

Doctor of Philosophy

By

Jack Paul Egerton

August 2017

Hydroacoustics as a tool for the assessment of fishes within existing and candidate marine protected areas (MPAs).

Hydroacoustics is widely used in fisheries research particularly for broad scale investigations such as fish stock assessments. There has, however, only been limited use of such technologies on a smaller scale for estimations of fish abundance, biomass and population size structures inside versus outside marine protected areas (MPAs). Further, within these, understanding the effects of habitat on fish distribution is of high importance especially in movements towards ecosystem based fisheries management (EBFM). The central aim of this study was to use hydroacoustic methods to determine fish distribution in and around existing MPAs and to identify areas which may be suitable for future protection. This was achieved through three field studies described in this thesis. The first study based at multiple locations across The Cayman Islands, examines in a spatial context the protection that MPAs afford to fish spawning aggregations (FSAs) of Nassau grouper (*Epinephelus striatus*) that they are designed to safeguard. The study found acoustic abundance estimates were similar to those made by divers. The results show the efficacy of hydroacoustic methods for FSA monitoring. The second study was centred on the Cabo Pulmo National Park (CPNP) in Baja California, Mexico. This study details the first application of hydroacoustics for MPA assessment. The results showed that fish abundance and biomass were significantly higher within the CPNP than nearby control areas. Further, the reefs within the CPNP had an order of magnitude greater abundance and biomass, demonstrating the importance of both habitat and protection for fish populations. The third field study was in the waters of the Qatari Gulf, where the hydroacoustic method was used to assess fish abundance values over different habitats to potentially determine areas suitable for protection. The results showed fish abundance, biomass and mean size were greatest over more complex habitats. The final data Chapter examines the data from the three field studies through a size spectra approach. This chapter details the first use of examining hydroacoustically derived fish size spectra in the marine environment. Fish size spectra was more curvilinear over more rugose habitats. This, in combination with examination of the slopes and heights of the spectra has shown that hydroacoustic size spectra approach may be of great value for rapidly assessing the status of fish communities in a non-destructive manner.

Declaration and Consent

Details of the Work

I hereby agree to deposit the following item in the digital repository maintained by Bangor University and/or in any other repository authorized for use by Bangor University.

Author Name: …………………………………………………………………..……………………………..

Title: ……………………………………………………………………..…………………………..

Supervisor/Department: …………………………………………………………………………………

Funding body (if any): ……………………………………………………………………………………

Qualification/Degree obtained: ………………………………………………………………………

This item is a product of my own research endeavours and is covered by the agreement below in which the item is referred to as "the Work". It is identical in content to that deposited in the Library, subject to point 4 below.

Non-exclusive Rights

Rights granted to the digital repository through this agreement are entirely non-exclusive. I am free to publish the Work in its present version or future versions elsewhere.

I agree that Bangor University may electronically store, copy or translate the Work to any approved medium or format for the purpose of future preservation and accessibility. Bangor University is not under any obligation to reproduce or display the Work in the same formats or resolutions in which it was originally deposited.

Bangor University Digital Repository

I understand that work deposited in the digital repository will be accessible to a wide variety of people and institutions, including automated agents and search engines via the World Wide Web.

I understand that once the Work is deposited, the item and its metadata may be incorporated into public access catalogues or services, national databases of electronic theses and dissertations such as the British Library's EThOS or any service provided by the National Library of Wales.

I understand that the Work may be made available via the National Library of Wales Online Electronic Theses Service under the declared terms and conditions of use

(http://www.llgc.org.uk/index.php?id=4676). I agree that as part of this service the National Library of Wales may electronically store, copy or convert the Work to any approved medium or format for the purpose of future preservation and accessibility. The National Library of Wales is not under any obligation to reproduce or display the Work in the same formats or resolutions in which it was originally deposited.

Statement 1:

This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree unless as agreed by the University for approved dual awards.

Signed …………………………………….... (candidate)

Date …………………………………………..

Statement 2:

This thesis is the result of my own investigations, except where otherwise stated. Where correction services have been used, the extent and nature of the correction is clearly marked in a footnote(s).

Other sources are acknowledged by footnotes giving explicit references. A bibliography is appended.

Signed ……………………………………… (candidate)

Date …………………………………………..

Statement 3:

I hereby give consent for my thesis, if accepted, to be available for photocopying, for inter-library loan and for electronic repositories, and for the title and summary to be made available to outside organisations.

Signed ……………………………………… (candidate)

Date …………………………………………..

NB: Candidates on whose behalf a bar on access has been approved by the Academic Registry should use the following version of **Statement 3:**

Statement 3 (bar):

I hereby give consent for my thesis, if accepted, to be available for photocopying, for inter-library loans and for electronic repositories after expiry of a bar on access.

Signed ………………………………………… (candidate)

Date ……………………………………………

Statement 4:

Choose **one** of the following options

a) I agree to deposit an electronic copy of my thesis (the Work) in the Bangor University (BU) Institutional Digital Repository, the British Library ETHOS system, and/or in any other repository authorized for use by Bangor University and where necessary have gained the required permissions for the use of third party material.	
b) I agree to deposit an electronic copy of my thesis (the Work) in the Bangor University (BU) Institutional Digital Repository, the British Library ETHOS system, and/or in any other repository authorized for use by Bangor University when the approved **bar on access** has been lifted.	
c) I agree to submit my thesis (the Work) electronically via Bangor University's e-submission system, however I **opt-out** of the electronic deposit to the Bangor University (BU) Institutional Digital Repository, the British Library ETHOS system, and/or in any other repository authorized for use by Bangor University, due to lack of permissions for use of third party material.	

Options B should only be used if a bar on access has been approved by the University.

In addition to the above I also agree to the following:

That I am the author or have the authority of the author(s) to make this agreement and do hereby give Bangor University the right to make available the Work in the way described above.

That the electronic copy of the Work deposited in the digital repository and covered by this agreement, is identical in content to the paper copy of the Work deposited in the Bangor University Library, subject to point 4 below.

That I have exercised reasonable care to ensure that the Work is original and, to the best of my knowledge, does not breach any laws – including those relating to defamation, libel and copyright.

That I have, in instances where the intellectual property of other authors or copyright holders is included in the Work, and where appropriate, gained explicit permission for the inclusion of that material in the Work, and in the electronic form of the Work as accessed through the open access digital repository, *or* that I have identified and removed that material for which adequate and appropriate permission has not been obtained and which will be inaccessible via the digital repository.

That Bangor University does not hold any obligation to take legal action on behalf of the Depositor, or other rights holders, in the event of a breach of intellectual property rights, or any other right, in the material deposited.

That I will indemnify and keep indemnified Bangor University and the National Library of Wales from and against any loss, liability, claim or damage, including without limitation any related legal fees and court costs (on a full indemnity bases), related to any breach by myself of any term of this agreement.

Signature: …………………………………………………… Date : ……………………………………………

Acknowledgements

Throughout this PhD, I've received vast amounts of support from all sorts of people from all over the world. Firstly however, a huge amount of thanks is due to both of my supervisors at Bangor; Professors John Turner and Lewis LeVay for great support throughout this PhD. Each of you provided the opportunity to conduct critical research for the success of this Thesis. John – many thanks for getting me involved with your Cayman Islands project. This has led to my first publication and was vital as a starting point in setting the scene for the thesis. Similarly Lewis for inviting me to join up with the Qatar project and helping me find financial assistance for conducting this research. Both of you saw the potential benefits and application of a methodology outside your areas of expertise and allowed me to have a shot at it. Whilst in Cayman I would particularly like to thank Croy McCoy for agreeing to the idea and helping making the project happen. Also thanks to Bradley Johnson, and Phillipe Bush, whose local knowledge and practical skills greatly assisted in making this a success. Further and important thanks goes to Laura Richardson, Sonja Bejarano who both made my trip out there so much fun.

In Mexico Dr Andrew Johnson of Scripps Institute of Oceanography, thanks heaps for helping set up that project mate. Thanks too for help throughout the seemingly endless manuscript revisions, and not so similarly - the surf trip in Baja! See you out in the states. Also Octavio Aburto-Oropeza for agreeing to flying me out there and for being so helpful. Tim Rowell from Scripps too – cheers mate and also Alfredo Girón for Tableau wizardry for the Datamares website. Also to all the crew from GCMP in La Paz who were so welcoming; muchas gracias amigos!

In Qatar, Lewis again and Dr Mark Walton. Thanks both for your support, manuscript reviews and for being such good company out in the Gulf. Thanks also goes to Professor Mike Kaiser and Dr Jan Geert Hiddink for helpful advice on size spectra analysis and statistics. Further, Professor Stuart Jenkins and Dr Katrien Van Landeghem for much help and advice in PhD meetings.

Last but not least, all my friends and especially my family for giving me huge support in this and in all of my endeavours, enduring positivity and for putting up with all my grumbling!

Contents

Table of Figures

Table of tables

16

1 Chapter 1. General Introduction

1.1 A background on fish hydroacoustics

Sound travels further through water than through air as opposed to light, and as such dolphins and toothed whales have evolved echo-sounding as an important sense (Wursig, 1989). The first reference to the utilisation of underwater sound by humans is from Leonardo Da Vinci in 1490, who noted that ships could be heard from great distances away through a long tube when placed under water (Urick, 1983). The speed of sound in water was first measured in 1826 by Colladon and Sturm but the first use of active echo sounding was by Maury (1859) in attempts to determine seabed depths (Fernandes et al. 2002). The Titanic disaster of 1912 lead to the first developments in using active sound to warn of navigational hazards – such as icebergs. Further developments in World War I lead to the development of the echosounder in efforts to locate submarines including the piezoelectric transducer by Langevin in 1917 (Simmonds and MacLennan, 2005). Echosounders began to be used to effectively determine bathymetry in 1937, and through this Alfred Wegener's 1915 theory of continental drift was confirmed through the discovery of the Mid-Atlantic ridge (Hess, 1962). The first fish detected by the use of echosounders was in 1927 when Rallier du Baty noted 'false echoes' resultant from fish schools off Newfoundland. Improvements in acoustic technology continued rapidly during World War II after which its potential was recognised for detecting fish (Simmonds and MacLennan, 2005). The first use of hydroacoustics to estimate fish abundance was in the late 1950's based on simple echo counting (Tungate, 1958) and valid echo integration methods were later developed by Scherbino and Truskanov (1966). Since then the technology, equipment and analyses have become increasingly sophisticated and useful in the field of fisheries research (Simmonds and MacLennan, 2005).

1.2 Hydroacoustics vs Underwater Visual Census (UVC)

In shallow water tropical environments, Underwater Visual Census (UVC) is probably the most common of the methods available for surveying fish communities (Irigoyen et al. 2013). The main advantage of the method is that it can provide high resolution species-specific information, and when fish lengths are estimated, biomass estimations are also possible. This species-specific information also allows any ecosystem shifts in the community structure to be discerned (Hughes, 1994) and can be used to calculate health indices (Aburto-Oropeza et al.

2011). It is also relatively inexpensive and is non-destructive, which is often a necessity when conducting research in Marine Protected Areas (MPAs) (Irigoyen et al. 2013). There are however also disadvantages with the UVC techniques, most importantly limited spatial coverage restricting diver surveys to detect changes in fish density across the scales which may comprise a contiguous reef community (Willis et al. 2000; Zenone et al. 2017). There can also be issues associated with underwater visibility (Harvey et al. 2002), high currents, increased operational risks at dawn and dusk, and diver depth and time limits (Sadovy de Mitcheson and Domeier 2005). There may also be a need to calibrate methods between different surveyors or groups of surveyors as research has shown that the type of UVC methodology chosen can lead to differences in the resultant data (Thresher and Gunn, 1986; Kulbicki and Sarramegna, 1999). UVC is also known to underestimate abundances of small fishes (Ackerman and Bellwood, 2003) and there are biases towards recording numbers of large fishes (Ward-Paige et al. 2010) and with the effects of diver avoidance by the fish community (Kulbicki 1998; Murphy and Jenkins 2010; Bozec et al. 2010). UVC surveys also only survey a limited volume of water in proximity to the seabed (MacNeil et al. 2008) and are therefore more focussed on demersal fishes rather than pelagic species.

Hydroacoustics has its own suite of biases and limitations for surveying fish communities but may also offer advantages in comparison to UVC (Zenone et al. 2017). Groundtruthing is the main issue in multispecies hydroacoustic fish surveys meaning species-specific information is lacking and this causes difficulties in establishing fish densities, lengths and biomass estimates (MacLennan and Simmonds 2005, Ryan et al. 2009). Throughout this thesis it is argued that, once this is acknowledged, the methodology can still provide highly useful information for which to monitor fish populations (Boswell et al. 2007). Similar to the effect of diver avoidance in UVC surveys, fish may also exhibit avoidance behaviour of the survey vessel (Draštík and Kubečka, 2005). The main advantages of hydroacoustic fish surveys are:

- They can cover a large spatial area in a short amount of time (Trenkel et al. 2011; Jones et al. 2012).
- There are no issues with water visibility (Gledhill et al. 1996; Zenone et al. 2017).
- There are no issues with diver depth limits or currents as experienced with UVC (Sadovy de Mitcheson and Domeier 2005).
- Most of the water column can be sampled (except zones close to surface and seabed) (Ona and Mitson 1996; Yurista et al. 2014).

- The technique avoids extraction or harm to fish populations.

- Raw data in high volumes are instantly recorded and retained for subsequent analyses (Trenkel et al. 2011).

- There is no size selectivity in the sampling of the fish community which is present with most fishing techniques (Wheeland and Rose, 2016).

- The same technique records data suitable for processing to give information on habitat type (Mackinson et al. 2004).

There have been few comparisons between hydroacoustics and UVC. In a recent study, Zenone et al. (2017), found that results between survey methods correlated statistically, but diver estimates were an order of magnitude higher. Taylor et al. (2006) conducting similar research to Chapter 3 in this thesis, reported similar acoustic density estimates to diver estimates of fish over their entire survey region, although total abundances differed which they concluded was likely due to differences in areal coverages and the patchy distribution of the fish.

1.3 Marine Protected Areas (MPAs) and fish abundance

Whilst the use of hydroacoustics is well established in fisheries research this is usually conducted on larger spatial scales than to be examined here. Further, most of this previous work is generally focussed on pelagic species, the distribution of which is not likely to be impacted as much by management effects than demersal species due to the larger ranges and less strong affiliations with bottom types (Claudet et al. 2008; Johnson et al. 2013). In this thesis, research studies include both the pelagic and demersal community and as these are all shallow-water tropical communities, the pelagic-demersal coupling is likely to be substantial (Bianchi et al. 2000).

MPAs are defined by the International Union of the Conservation of Nature (IUCN) as '*Any area of intertidal or subtidal terrain, together with its overlying waters and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment*" (IUCN, 2008). MPAs have long been known as an effective tool to increase fish abundance by reducing the pressures caused by fishing (e.g. Beverton and Holt, 1957). MPA effects have been the focus of considerable research and meta-analyses report increases in fish abundance, biomass, diversity (Lester et al. 2009; Sciberras et al. 2013, Starr et al. 2015) and also through larger mean sizes of fishes (Edgar and Stuart-Smith, 2009). The majority of studies on MPA effects on fish communities have been

conducted via UVC surveys (see below). There have been other studies using methods such as Baited Remote Underwater Video (BRUVs) (Malcolm et al. 2007), and fishing methods (Sweeting et al. 1999), and recently passive acoustics (Piercy et al. 2014). There has also been research using active hydroacoustics within MPAs, however the focus has been on comparability between methods (e.g. Rudershausen et al. 2010).

1.4 Effects of substrate on fish distribution

The relationship between marine fishes and their habitats is important for the understanding of their distributions (van der Kooij et al. 2011), and it should be noted that throughout this thesis the word 'habitat' is used synonymously with 'substrate' (Diaz et al. 2004). As with the effects of management mentioned above, there has been little use of hydroacoustics to examine habitat effects. This lack of research may be largely due to the presence of the 'acoustic deadzone' (Ona and Mitson, 1996), where fishes in close proximity to the seabed are not possible to survey. Once this is acknowledged, however, hydroacoustics can still provide valuable data on the demersal community if adequately separated from the substrate, in addition to the pelagic community and how it is affected by substrate/habitat type *inter alia*. The approach of using hydroacoustics to examine demersal species habitat use has been underutilised in the literature. Boswell et al (2007) used hydroacoustics to reveal fish preferential habitat use of shell habitat in a shallow estuarine environment. Kracker (2007) also used hydroacoustics to examine differences in fish abundances over different habitats in Gray's Reef National Marine Sanctuary (GRNMS), finding that mixed habitat had higher fish density and biomass than the sand habitat. The methods have also been conducted to examine fish use of habitat in the freshwater systems, e.g. Lian et al. (2017), who showed the influence of macrophytes on fish distribution. The effects of habitat on demersal species have however long been examined using other survey methodologies such as UVC. Most of these studies have found increasing habitat complexity results in increases in fish abundance (Risk, 1972; Luckhurst and Luckhurst, 1978; Komyakova et al. 2013; Graham and Nash, 2013), and also biomass (Grigg, 1994). A more complex substratum provides more habitat for invertebrate species which serve as food resources for fishes (Parrish et al. 1985). Greater structural complexity also provides more areas for shelter for fishes, resulting in the higher abundances (Coles and Tarr, 1990). Further, differences between habitat types may be seen with different classes of fishes (Wilson et al. 2010), where a higher abundance of small fish is often revealed in more complex habitats (Graham et al. 2007; Wilson et al. 2010). The abundance of live coral in an area has also been

seen to have a positive effect on fish abundance (Carpenter et al. 1981; Bell and Galzin, 1984; Bouchon-Navro and Bouchon, 1989, Graham and Nash, 2013).

1.5 Size spectra

The size spectra of a fish community describes how the total abundance or biomass varies between fish size classes, allowing the relative numbers of larger vs smaller individuals to be quantitatively examined (Edwards et al. 2017). The concept of size spectra relies upon the theory of 'pyramid of numbers' first described by Elton (1927) whereby numerous small species get eaten by ever fewer numbers of predators with increases in trophic levels (Sprules and Barth 2015). Size spectra approaches then lay dormant until 1967 when they were re-examined for use in examining plankton (Sheldon and Parsons, 1967; Parsons et al. 1969). The research continued from there and Sheldon and Kerr (1972), eloquently demonstrated how it could be applied to estimate the potential abundance of top predators (the Monsters) in Loch Ness. The methodology flips a trophic pyramid on its side so abundance is on the y axis and size class on the x, both axes are logged and as a result a slope can be visualised (Treblico et al. 2013) (Figure 1.1). The steepness of this slope gives an indication of the relative abundances of small to large fish (Jennings, 2005). Through this the effects of fishing can be discerned, with more large fish removed steepening the slope (Pope and Knights 1982; Blanchard et al. 2009; Robinson et al. 2016). Further, the intercept and midpoint height of the slope gives information on overall productivity and abundance of the community, respectively (Bianchi et al. 2000; Daan et al. 2005; Sweeting et al. 2009).
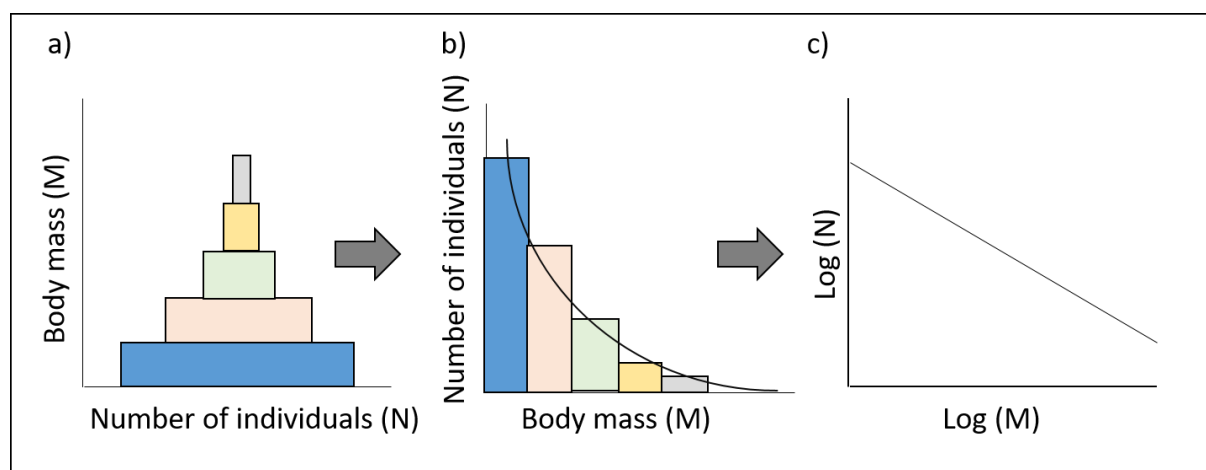
Figure 1.1. The principle behind deriving size spectra (adapted from Treblico et al. 2013). This shows how a trophic pyramid is flipped on its side (a – b) and then logged so that a linear slope is presented (c).

Managers are shifting focus from single species assessments to more ecosystem wide approaches (Bianchi et al. 2000; Trenkel et al. 2011), with the progression towards an Ecosystem Based Approach to Fisheries (EBAF) (Garcia et al. 2003). In this context using size based analyses may be a valuable tool to determine changes in fish communities (Shin et al. 2005; Jennings, 2005; Wilson et al. 2010; Blanchard et al. 2017). Size spectra based analyses derived from acoustic data have been highlighted as potentially having much merit to examine changes in fish communities (Trenkel et al. 2011). Without any issues due to the size selectivity problems of fishing gear (Shin et al. 2005), acoustic approaches may be able to show a more complete picture of the fish ecosystem (Yurista et al. 2014). Without the size selective bias present with other survey methodologies, which may under sample small fish, this approach revealed that size spectra may fit quadratic functions better than linear functions, which has been highlighted as an open question (Shin and Cury, 2004). Additionally, survey instruments such as hydroacoustics can provide greater spatial and temporal resolution in comparison to traditional methods (Sprules and Barth, 2015). Hydroacoustic size spectra have been examined in lacustrine environments (e.g. Brandt et al. 1991; Yule et al. 2013; Yurista et al. 2014, Pollom and Rose, 2015, de Kerckhove et al. 2015; Wheeland and Rose, 2016), but there is no published research on deriving size spectra from hydroacoustic methods in the marine environment.

1.6 Thesis hypotheses

In this thesis three case studies sites are used in order to test the efficacy of acoustics in for the purpose of examining how hydroacoustics can be used to assess Marine Protected Areas (MPAs) and candidate MPA areas. These study locations were: The Cayman Islands, Mexico and Qatar and in these study areas the following hypotheses were tested:

1. There will be no significant difference between hydroacoustic fish abundances and those provided by SCUBA divers at Fish Spawning Aggregations (FSAs) in The Cayman Islands.
2. Hydroacoustics will be able to a) locate FSAs in historic areas where it is unknown if they still occur and b) determine if FSAs are afforded adequate spatial protection.
3. There will be no significant difference between hydroacoustic fish densities, biomass and mean size in comparison to those provided by SCUBA divers at reef sites in The Cabo Pulmo National Park in Mexico.
4. There will be greater abundances of fishes in areas that are protected from fishing in comparison to fished control sites.

5. a) More rugose habitats will have significantly higher fish density and biomass and mean size than less rugose sites and b) the fishes detected will have a greater association (proximity) with the seabed.

6. Hydroacoustic data using variables of abundance and size (derived from Target Strength) analysed through a Normalised Biomass Size Spectra (NBSS) approach will help describe the effects of protection and habitat type on the fish distribution. Specifically, a) protected areas will have shallower slopes, and b) intercepts and midpoint heights will be higher over more rugose habitats.

7. Without any size selective bias in the hydroacoustic data, NBSS will fit quadratic functions significantly better than linear functions.

1.7   Thesis objectives

The overall aim of this thesis is to examine how hydroacoustics can be applied to help provide data to assist managers with fish assessment for MPAs. This is examined in three locations: The Cayman Islands, in the Qatari waters of the Arabian Gulf and at The Cabo Pulmo National Park in Mexico. The hydroacoustic data from these case studies are then examined together through a size spectra based approach.

1) The first overall objective in order to test hypotheses 1 and 2, is to compare hydroacoustic results with other data sources i.e. Underwater Visual Census (UVC) SCUBA surveys. This is conducted at FSA sites in The Cayman Islands and at reef sites in the CPNP.

2) Following this, to test hypotheses 3, 4 and 5 the hydroacoustic fish distribution data are examined spatially to investigate how such data can provide valuable information to managers. This compares FSA location to that of the protected areas designed to encompass them in the Cayman Islands (hypothesis 3), the effects of protection and habitat on fish distribution at the CPNP and control sites (hypotheses 4 and 5a), and also between different habitat types in Qatar (hypotheses 5a and b).

3) The final objective in order to test hypotheses 6 and 7, is to examine the hydroacoustic data gained from the previous three chapters in terms of size spectra to examine the additional information that such an approach can provide. Specifically all sites with be statistically tested as to whether the data better fits quadratic vs linear models. Further, from this approach additional information is therefore also given relevant to hypotheses 4 and 5a.

1.8 Thesis structure

A central question of this thesis is how hydroacoustics can be used to examine effects of management or fishing on the abundances of fishes. Whilst there has been suggestion that hydroacoustics could be a valuable way to monitor MPAs (Kracker 2007; Polunin et al. 2009), the research detailed in Chapter 4 of this thesis is possibly the first attempt to use hydroacoustics to determine MPA effects on a fish community (search terms "MPA" and "hydroacoustics" in Google Scholar and Elsevier). Chapter 3 also uses hydroacoustics to examine fish within MPAs, but here with the focus being on FSAs, the approach is different placing the aggregations in a spatial context rather than an inside vs outside reserve comparison. Chapter 5, although not examining current protection is also relevant in that the research was conducted from an angle of how hydroacoustics can be used to identify areas/habitats suitable for future protection. Chapter 6 in this thesis may be the first study examining hydroacoustically derived size spectra in the marine environment (search terms "size spectra" "marine" and "hydroacoustics" in Google Scholar and Elsevier). The chapter uses data from the previous three chapters to further investigate the acoustic data processing options available, the effects of marine protection and also habitat on resultant size spectra.

The following chapters have been published in peer reviewed journals, or are in review. They appear in a similar but slightly reduced version to that provided in this thesis:

Egerton, J.P., Johnson, A.F., Le Vay, L., McCoy, C.M., Semmens, B.X., Heppell, S.A. and Turner, J.R., 2017. Hydroacoustics for the discovery and quantification of Nassau grouper (*Epinephelus striatus*) spawning aggregations. **Coral Reefs**, 36(2), pp.589-600.

Chapter 4. Egerton, J. P., Johnson, A.F., Turner, J., Le Vay, L., Mascareñas-Osorio, I., Aburto-Oropeza, O. Hydroacoustics as a tool to examine the effects of protection and habitat type on marine fish communities. *Accepted, In press:* **Scientific Reports**

Chapter 5. Egerton, J. P. Mohsin Al-Ansi, Mohamed Abdallah, Walton, M., Hayes, J., Turner, J., Al-Maslamani,I., Mohannadi, M., Le Vay, L. Hydroacoustics to examine fish association with shallow offshore habitats in the Arabian Gulf. *Accepted, In press:* **Fisheries Research**

These articles were all written by the author of this thesis and co-author contributions were mainly restricted to supervision, organisation and the provision of comments prior to submission. In the corresponding relevant chapters the 'we' refers to these publication co-authors.

1.9   References

Aburto-Oropeza, O., Erisman, B., Galland, G.R., Mascareñas-Osorio, I., Sala, E. and Ezcurra, E., 2011. Large recovery of fish biomass in a no-take marine reserve. *PLoS One*, *6*(8), p.e23601.

Ackerman, J.L. and Bellwood, D.R., 2000. Reef fish assemblages: a re-evaluation using enclosed rotenone stations. *Marine Ecology Progress Series*, *206*, pp.227-237.

Bell, J.D. and Galzin, R., 1984. Influence of live coral cover on coral-reef fish communities. *Marine Ecology Progress Series*, pp.265-274.

Beverton, R.J. and Holt, S.J., 1957. *On the dynamics of exploited fish populations* (Vol. 11). Springer Science and Business Media.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Jin, X., Koranteng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F. and Zwanenburg, K., 2000. Impact of fishing on size composition and diversity of demersal fish communities. *ICES Journal of Marine Science*, *57*(3), pp.558-571.

Blanchard, J.L., Jennings, S., Law, R., Castle, M.D., McCloghrie, P., Rochet, M.J. and Benoît, E., 2009. How does abundance scale with body size in coupled size-structured food webs?. *Journal of Animal Ecology*, *78*(1), pp.270-280.

Blanchard, J.L., Heneghan, R.F., Everett, J.D., Trebilco, R. and Richardson, A.J., 2017. From bacteria to whales: using functional size spectra to model marine ecosystems. *Trends in Ecology and Evolution*.

Boswell, K.M., Wilson, M.P. and Wilson, C.A., 2007. Hydroacoustics as a tool for assessing fish biomass and size distribution associated with discrete shallow water estuarine habitats in Louisiana. *Estuaries and Coasts*, *30*(4), pp.607-617.

Bouchon-Navaro, Y. and Bouchon, C., 1989. Correlations between chaetodontid fishes and coral communities of the Gulf of Aqaba (Red Sea). *Environmental Biology of Fishes*, *25*(1), pp.47-60.

Bozec, Y.M., Kulbicki, M., Laloë, F., Mou-Tham, G. and Gascuel, D., 2011. Factors affecting the detection distances of reef fish: implications for visual counts. *Marine Biology*, *158*(5), pp.969-981.

Brandt, S.B., Mason, D.M., Patrick, E.V., Argyle, R.L., Wells, L., Unger, P.A. and Stewart, D.J., 1991. Acoustic measures of the abundance and size of pelagic planktivores in Lake Michigan. *Canadian Journal of Fisheries and Aquatic Sciences*, *48*(5), pp.894-908.

Carpenter, K.E., Miclat, R.I., Albaladejo, V.D. and Corpuz, V.T., 1981, May. The influence of substrate structure on the local abundance and diversity of Philippine reef fishes. In *Proceedings of the 4th International Coral Reef Symposium* (Vol. 2, pp. 497-502).

Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., García-Charton, J.A., Pérez-Ruzafa, Á., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F. and Culioli, J.M., 2008. Marine reserves: size and age do matter. *Ecology letters*, *11*(5), pp.481-489.

Coles, S.L. and Tarr, B.A., 1990. Reef fish assemblages in the western Arabian Gulf: a geographically isolated population in an extreme environment. *Bulletin of Marine Science*, *47*(3), pp.696-720.

Daan, N., Gislason, H., G. Pope, J. and C. Rice, J., 2005. Changes in the North Sea fish community: evidence of indirect effects of fishing?. *ICES Journal of marine Science*, *62*(2), pp.177-188.

de Kerckhove, D.T., Shuter, B.J. and Milne, S., 2015. Acoustically derived fish size spectra within a lake and the statistical power to detect environmental change. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.565-574.

Diaz, R.J., Solan, M. and Valente, R.M., 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management*, *73*(3), pp.165-181.

Draštík, V., Kubečka, J., Čech, M., Frouzová, J., Říha, M., Juza, T., Tušer, M., Jarolím, O., Prchalová, M., Peterka, J. and Vašek, M., 2009. Hydroacoustic estimates of fish stocks in temperate reservoirs: day or night surveys?. *Aquatic Living Resources*, *22*(1), pp.69-77.

Edgar, G.J. and Stuart-Smith, R.D., 2009. Ecological effects of marine protected areas on rocky reef communities—a continental-scale analysis. *Marine Ecology Progress Series*, *388*, pp.51-62.

Edwards, A.M., Robinson, J.P., Plank, M.J., Baum, J.K. and Blanchard, J.L., 2017. Testing and recommending methods for fitting size spectra to data. *Methods in Ecology and Evolution*, *8*(1), pp.57-67.

Elton, C.S., 1927. Animal ecology. University of Chicago Press.

Fernandes, P.G., Gerlotto, F., Holliday, D.V., Nakken, O. and Simmonds, E.J., 2002. Acoustic applications in fisheries science: the ICES contribution. ICES. *ICES Marine Science Symposia*, 215: 483–492.

Garcia, S.M., 2003. *The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook* (No. 443). Food and Agriculture Org.

Gledhill, C.T., Lyczkowski-Shultz, J., Rademacher, K., Kargard, E., Crist, G. and Grace, M.A., 1996. Evaluation of video and acoustic index methods for assessing reef-fish populations. *ICES Journal of Marine Science*, *53*(2), pp.483-485.

Graham, N.A.J. and Nash, K.L., 2013. The importance of structural complexity in coral reef ecosystems. *Coral Reefs*, *32*(2), pp.315-326.

Graham, N.A., Wilson, S.K., Jennings, S., Polunin, N.V., Robinson, J.A.N., Bijoux, J.P. and Daw, T.M., 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. *Conservation biology*, *21*(5), pp.1291-1300.

Grigg, R.W., 1994. Effects of sewage discharge, fishing pressure and habitat complexity on coral ecosystems and reef fishes in Hawaii. *Marine Ecology Progress Series*, pp.25-34.

Hess, H.H., 1962. History of ocean basins. *Petrologic studies (Buddington volume)*, pp.599-620.

Hughes, T.P., 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science-AAAS-Weekly Paper Edition*, *265*(5178), pp.1547-1551.

Irigoyen, A.J., Galván, D.E., Venerus, L.A. and Parma, A.M., 2013. Variability in abundance of temperate reef fishes estimated by visual census. *PLoS One*, *8*(4), p.e61072.

IUCN World Commission on Protected Areas (IUCN-WCPA). 2008. Washington, D.C.: IUCN-WCPA, National Oceanic and Atmospheric Administration and The Nature Conservancy.

Jennings, S., 2005. Size-based analyses of aquatic food webs. Aquatic food webs: an ecosystem approach. Belgrano, A., Scharler, UM, Dunne, J. and Ulanowicz, RE. Oxford University Press, Oxford, pp.86-97

Jones, D.T., Wilson, C.D., De Robertis, A., Rooper, C.N., Weber, T.C. and Butler, J.L., 2012. Evaluation of rockfish abundance in untrawlable habitat: combining acoustic and complementary sampling tools. *Fishery Bulletin*, *110*(3), pp.332-343.

Johnson, A.F., Jenkins, S.R., Hiddink, J.G. and Hinz, H., 2013. Linking temperate demersal fish species to habitat: scales, patterns and future directions. *Fish and fisheries*, *14*(3), pp.256-280.

Komyakova, V., Munday, P.L. and Jones, G.P., 2013. Relative importance of coral cover, habitat complexity and diversity in determining the structure of reef fish communities. *PLoS One*, *8*(12), p.e83178.

Kracker, L., 2007. Hydroacoustic surveys: A non-destructive approach to monitoring fish distributions at National Marine Sanctuaries. NOAA Technical Memorandum NOS NCCOS 66

Kulbicki, M., 1998. How the acquired behaviour of commercial reef fishes may influence the results obtained from visual censuses. *Journal of Experimental Marine Biology and Ecology*, *222*(1), pp.11-30.

Kulbicki, M. and Sarramégna, S., 1999. Comparison of density estimates derived from strip transect and distance sampling for underwater visual censuses: a case study of Chaetodontidae and Pomacanthidae. *Aquatic Living Resources*, *12*(5), pp.315-325..

Lester, S.E., Halpern, B.S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B.I., Gaines, S.D., Airamé, S. and Warner, R.R., 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, *384*, pp.33-46.

Lian, Y., Huang, G., Godlewska, M., Cai, X., Li, C., Ye, S., Liu, J. and Li, Z., 2017. Hydroacoustic estimates of fish biomass and spatial distributions in shallow lakes. *Chinese Journal of Oceanology and Limnology*, pp.1-11.

Luckhurst, B.E. and Luckhurst, K., 1978. Analysis of the influence of substrate variables on coral reef fish communities. *Marine Biology*, *49*(4), pp.317-323.

Mackinson, S., Freeman, S., Flatt, R. and Meadows, B., 2004. Improved acoustic surveys that save time and money: integrating fisheries and ground-discrimination acoustic technologies. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.129-140.

MacNeil, M.A., Tyler, E.H., Fonnesbeck, C.J., Rushton, S.P., Polunin, N.V. and Conroy, M.J., 2008. Accounting for detectability in reef-fish biodiversity estimates. *Marine Ecology Progress Series*, *367*, pp.249-260.

Malcolm, H.A., Gladstone, W., Lindfield, S., Wraith, J. and Lynch, T.P., 2007. Spatial and temporal variation in reef fish assemblages of marine parks in New South Wales, Australia—baited video observations. *Marine Ecology Progress Series*, *350*, pp.277-290.

Maury, M.F., 1859. The Physical Geography of the Sea. Thomas Nelson, London. 493 pp.

Murphy, H.M. and Jenkins, G.P., 2010. Observational methods used in marine spatial monitoring of fishes and associated habitats: a review. *Marine and Freshwater Research*, *61*(2), pp.236-252.

Ona, E. and Mitson, R.B., 1996. Acoustic sampling and signal processing near the seabed: the deadzone revisited. *ICES Journal of Marine Science*, *53*(4), pp.677-690.

Parsons, T.R., 1969. The use of particle size spectra in determining the structure of a plankton community. *Journal of the Oceanographical Society of Japan*, *25*(4), pp.172-181.

Parrish, J.D., Callahan, M.W. and Norris, J.E., 1985, May. Fish trophic relationships that structure reef communities. In *Proceedings of the 5th International Coral Reef Congress* (Vol. 4, pp. 73-78).

Piercy, J.J., Codling, E.A., Hill, A.J., Smith, D.J. and Simpson, S.D., 2014. Habitat quality affects sound production and likely distance of detection on coral reefs. *Marine Ecology Progress Series*, *516*, pp.35-47.

Pollom, R.A. and Rose, G.A., 2015. Size-based hydroacoustic measures of within-season fish abundance in a boreal freshwater ecosystem. *PloS One*, *10*(4), p.e0124799.

Polunin, N., Bloomfield, H., Sweeting, C. and McCandless, D., 2009. Developing indicators of MPA effectiveness: finfish abundance and diversity in a Yorkshire prohibited trawl area. *Report to MFA, London*.

Pope, J.G. and Knights, B.J., 1982. Comparison of length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *Multispecies Approaches to Fisheries Management Advice*, *59*, pp.116-118.

Rallier du Baty, R. 1927. La pêche sur le banc de Terre-Neuve et autour des îles Saint-Pierre et Miquelon. Office Scientifique et Technique des Pêches Maritimes. Mémoires (Série Spéciale), 7. 142 pp

Risk, M.J., 1972. *Fish diversity on a coral reef in the Virgin Islands*. Smithsonian Institution.

Robinson, J.P., Williams, I.D., Edwards, A.M., McPherson, J., Yeager, L., Vigliola, L., Brainard, R.E. and Baum, J.K., 2017. Fishing degrades size structure of coral reef fish communities. *Global Change Biology*, *23*(3), pp.1009-1022.

Rudershausen, P.J., Mitchell, W.A., Buckel, J.A., Williams, E.H. and Hazen, E., 2010. Developing a two-step fishery-independent design to estimate the relative abundance of deepwater reef fish: application to a marine protected area off the southeastern United States coast. *Fisheries Research*, *105*(3), pp.254-260..

Ryan, T.E., Kloser, R.J. and Macaulay, G.J., 2009. Measurement and visual verification of fish target strength using an acoustic-optical system attached to a trawlnet. *ICES Journal of Marine Science*, *66*(6), pp.1238-1244.

Sadovy de Mitcheson, Y. and Domeier, M., 2005. Are aggregation-fisheries sustainable? Reef fish fisheries as a case study. *Coral Reefs*, *24*(2), pp.254-262.

Scherbino, M. and Truskanov, M.D., 1966. Determination of fish concentration by means of acoustic apparatus. *ICES Journal of Marine Science*, *3*, pp.6.

Sciberras, M., Jenkins, S.R., Mant, R., Kaiser, M.J., Hawkins, S.J. and Pullin, A.S., 2015. Evaluating the relative conservation value of fully and partially protected marine areas. *Fish and Fisheries*, *16*(1), pp.58-77.

Sheldon, R.W. and Kerr, S.R., 1972. The population density of monsters in Loch Ness. *Limnology and Oceanography*, 17(5), pp.796-798.

Sheldon, R.W. and Parsons, T.R., 1967. *A practical manual on the use of the Coulter Counter in marine research*. Coulter Electronics Sales.

Shin, Y.J., Rochet, M.J., Jennings, S., Field, J.G. and Gislason, H., 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. *ICES Journal of Marine Science*, *62*(3), pp.384-396.

Shin, Y.J. and Cury, P., 2004. Using an individual-based model of fish assemblages to study the response of size spectra to changes in fishing. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(3), pp.414-431.

Simmonds, J. and MacLennan, D.N., 2005. *Fisheries acoustics: theory and practice*. John Wiley and Sons.

Sprules, W.G. and Barth, L.E., 2015. Surfing the biomass size spectrum: some remarks on history, theory, and application. *Canadian Journal of Fisheries and Aquatic Sciences*, 73(4), pp.477-495.

Starr, R.M., Wendt, D.E., Barnes, C.L., Marks, C.I., Malone, D., Waltz, G., Schmidt, K.T., Chiu, J., Launer, A.L., Hall, N.C. and Yochum, N., 2015. Variation in responses of fishes across multiple reserves within a network of marine protected areas in temperate waters. *PloS One*, *10*(3), p.e0118502.

Sweeting, C.J., Badalamenti, F., D'Anna, G., Pipitone, C. and Polunin, N.V.C., 2009. Steeper biomass spectra of demersal fish communities after trawler exclusion in Sicily. *ICES Journal of Marine Science*, *66*(1), pp.195-202.

Taylor, J.C., Eggleston, D.B. and Rand, P.S.. 2006. Nassau grouper (*Epinephelus striatus*) spawning aggregations: hydroacoustic surveys and geostatistical analysis. In: Emerging technologies for reef fisheries research and management. Seattle, WA, NOAA, pp. 18-25. (NOAA Professional Paper NMFS, 5)

Thresher, R.E. and Gunn, J.S., 1986. Comparative analysis of visual census techniques for highly mobile, reef-associated piscivores (Carangidae). *Environmental Biology of Fishes*, *17*(2), pp.93-116.

Trebilco, R., Baum, J.K., Salomon, A.K. and Dulvy, N.K., 2013. Ecosystem ecology: size-based constraints on the pyramids of life. *Trends in Ecology and Evolution*, *28*(7), pp.423-431.

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C., 2011. Underwater acoustics for ecosystem-based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, *442*, pp.285-301.

Tungate, D.S. 1958. Echo-sounder surveys in the autumn of 1956. *Fishery Investigations*, Series 2, 12: 3–17

Urick, R.J. 1983. Principles of Underwater Sound. McGraw Hill, New York. 423 pp

van der Kooij, J., Kupschus, S. and Scott, B.E., 2011. Delineating the habitat of demersal fish assemblages with acoustic seabed technologies. *ICES Journal of Marine Science*, *68*(9), pp.1973-1985.

Ward-Paige, C., Flemming, J.M. and Lotze, H.K., 2010. Overestimating fish counts by non-instantaneous visual censuses: consequences for population and community descriptions. *PLoS One*, *5*(7), p.e11722.

Wheeland, L.J. and Rose, G.A., 2015. Acoustic measures of lake community size spectra. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.557-564.

Willis, T.J., Millar, R.B. and Babcock, R.C., 2000. Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. *Marine Ecology Progress Series*, pp.249-260.

Wilson, S.K., Fisher, R., Pratchett, M.S., Graham, N.A.J., Dulvy, N.K., Turner, R.A., Cakacaka, A. and Polunin, N.V.C., 2010. Habitat degradation and fishing effects on the size structure of coral reef fish communities. *Ecological Applications*, *20*(2), pp.442-451.

Wursig, B., 1989. Cetaceans. *Science*, *244*(4912), pp.1550-1558.

Yule, D.L., Adams, J.V., Hrabik, T.R., Vinson, M.R., Woiak, Z. and Ahrenstorff, T.D., 2013. Use of classification trees to apportion single echo detections to species: application to the pelagic fish community of Lake Superior. *Fisheries research*, *140*, pp.123-132.

Yurista, P.M., Yule, D.L., Balge, M., VanAlstine, J.D., Thompson, J.A., Gamble, A.E., Hrabik, T.R., Kelly, J.R., Stockwell, J.D. and Vinson, M.R., 2014. A new look at the Lake Superior biomass size spectrum. *Canadian Journal of Fisheries and Aquatic Sciences*, *71*(9), pp.1324-1333.

Zenone, A.M., Burkepile, D.E. and Boswell, K.M., 2017. A comparison of diver vs. acoustic methodologies for surveying fishes in a shallow water coral reef ecosystem. *Fisheries Research*, *189*, pp.62-66.

2   Chapter 2. Hydroacoustic methods

2.1   Hydroacoustic theory

Throughout this thesis analyses are based upon echo integration (EI), (also known as Sv/TS scaling) in Sonar5 (ver 6.0.4) software (Balk and Lindem, 2000). This method calculates fish abundance by dividing the average reflection from all fish over a segment (the volume backscattering coefficient, sv with units of $m^2 \, m^{-3}$) by the average Target Strength (TS) from individual fish (Winfield et al. 2011) which is derived from the average backscattering cross section (σbs) (Draštík et al. 2009). The average backscattering cross section (σ_{bs}) is the acoustic reflectivity of the target and is defined by the following relationship:

$$\sigma_{bs} = R^2 Ib/Ii$$

where R is the range, Ib is the intensity of the backscattered pulse and Ii is the intensity of the incident pulse.

TS is derived from the average backscattering cross section (σbs) but is in the logarithmic units of decibels (dB) and they therefore relate in the following manner:

$$TS = 10 \log 10 \, (\sigma_{bs}),$$

$$\text{or reformulated as: } \sigma_{bs} = 10^{\wedge}(TS/10)$$

The total volume backscattering sv is the fish density multiplied by the average backscattering cross section (σbs) within the sampled volume:

$$sv = \rho(\sigma_{bs}) \text{ where } \rho \text{ is fish density.}$$

$$\text{Or as: } sv = \Sigma \, \sigma bs/V$$

where V is the sampled volume (Simmonds and MacLennan, 2005).

The area scattering coefficient sA (in m^2/ha) is derived from Sv, it is defined as the integral of sv with respect to depth (Simmonds and MacLennan, 2005).

$$sA = \int_{z1}^{z2} sv * dz$$

where z1 is the upper depth, z2 the lower depth and dz the total depth of the layer.

This can then also be used to provide density of fish per ha:

Fish per ha $=sA/\sigma_{bs}$

There is a linear relationship between sA and fish biomass (Simmonds and MacLennan, 2005), and it can therefore be used as a proxy for this (Boswell et al. 2007).

More specific details on the methods used are given in the methods sections of each data chapter of this thesis.

2.2 Equipment specifications

In all the chapters detailed in this thesis a Biosonics DTX Split beam 200 kHz echosounder was used for the surveys (Figure 2.1). The transducer was pole mounted over the side of the survey vessels as close to the centre of roll and pitch as possible, with the transducer face 1m below the surface (Figure 2.2). Acoustic data were georeferenced with an integrated Garmin 17Xhvs GPS, and collected with Biosonics acquisition software (Visual Acquisition 6). The circular transducer has a beam opening angle of 6.8º. Pulse duration was 0.4ms and the specified ping rate was 10 per second. Calibration of the echosounder occurred before the start of all the surveys using a Biosonics 36mm Tungsten Carbide 200 kHz Calibration Sphere following the standard methods of Foote et al (1987).

Figure 2.1 Biosonics DTX Split beam echosounder (200 kHz) used for all the surveys in this thesis.

Figure 2.2 Example of how the echosounder transducer was pole mounted over the side of the survey vessels.

An echosounder works by transmitting a pulse of sound (a 'ping') down through the water column. This ping reflects off any particle in the water column and also the seabed and returns a component of the reflected sound back to the transducer. With fish, the reflected sound is largely due to the size of the swimbladder which is proportional to fish length. Other factors are however important in the strength of the returning echo, especially the tilt angle of the fish. Being 'split beam' means that the acoustic beam is split into four quadrants which means the movement of fish through the beam can be tracked as it moves from one segment to another. The data are initially processed by the surface unit and is recorded and shown as an 'echogram' (Figure 2.3) on a laptop computer connected to the surface unit via an Ethernet cable.
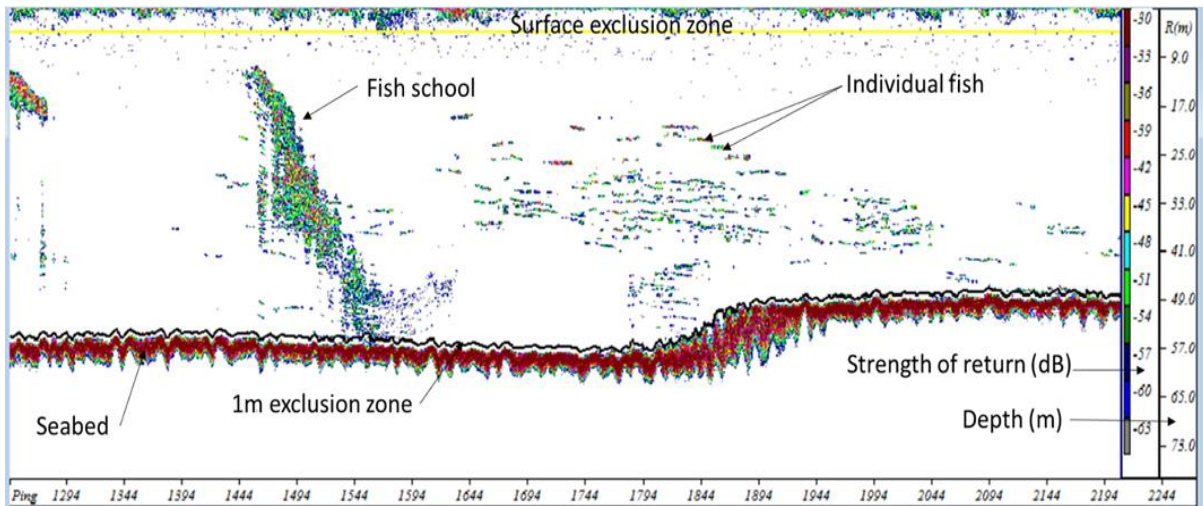
Figure 2.3 Example echogram in Sonar5 (ver 6.0.4) showing how the acoustic data is displayed in the processing software. Strength of signal return (dB) and depth are shown on the y axis whilst ping number is shown on the y axis.

Groundtruthing has been known as the Achilles heel of fish acoustics (Mackinson et al. 2004). It is needed to apportion echoes to fish species in order to allow accurate conversions of TS-Length and subsequent biomass values. This is extremely challenging in a multispecies situation such as described here in the Mexico and Qatar surveys (Chapters 4 and 5). In all surveys however this was attempted via the use of self-made bespoke pelagic towed camera systems capable of deployment to around 30m. (Figure 2.4, Figure 2.5). The systems proved extremely useful in Chapter 3, when it was used to confirm when a fish aggregation was comprised of a species other than the target species (Nassau grouper) (Figure 2.6, Figure 2.7).
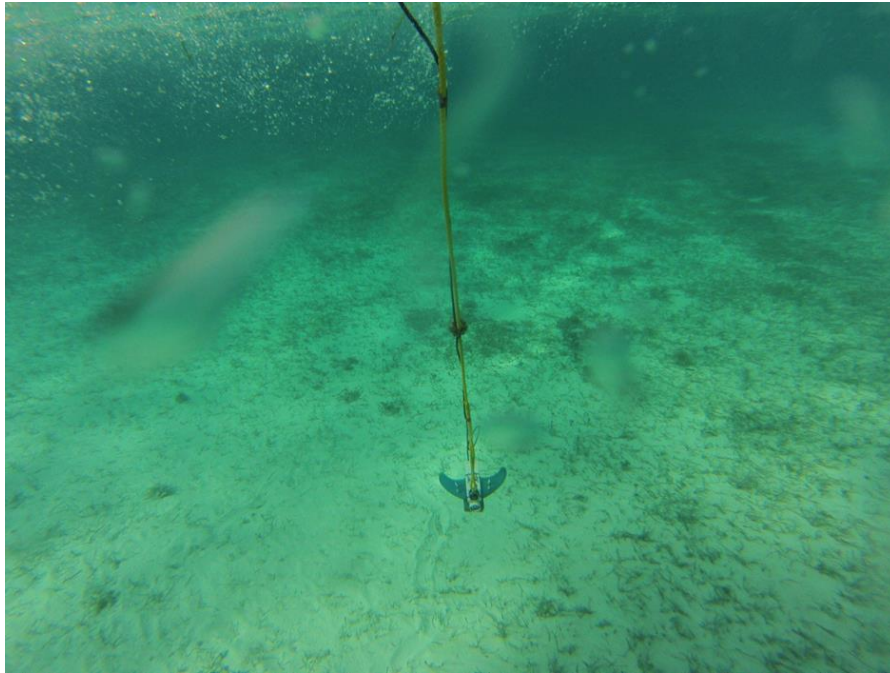
Figure 2.4 Towed camera system using a Sony 37CSHR camera with a live surface feed used to determine the species comprising fish aggregations in The Cayman Islands (Chapter 2).
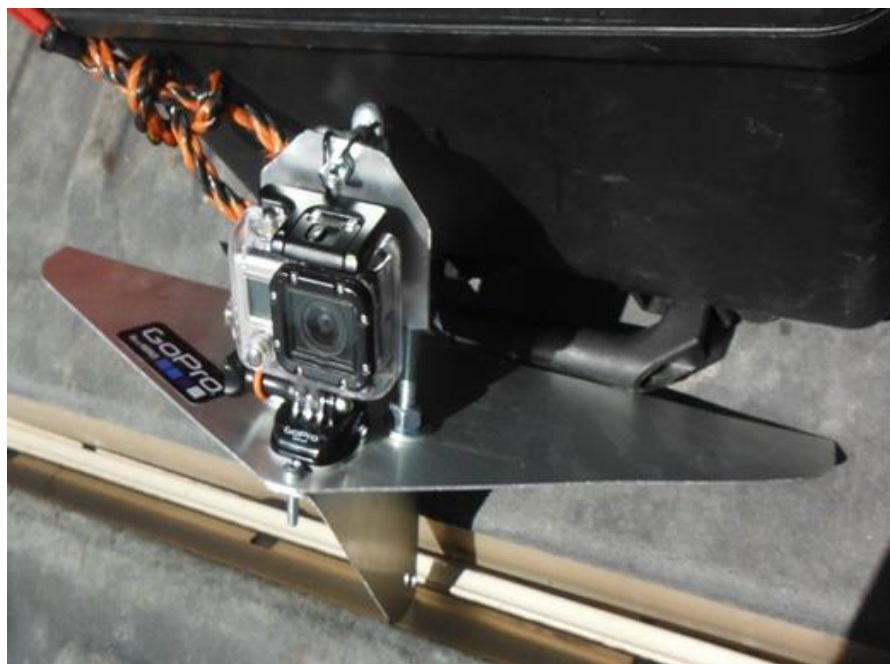
Figure 2.5 Towed camera system using a GoPro Hero4 Black camera, used for groundtruthing habitat and fish species in Mexico (Chapter 3).
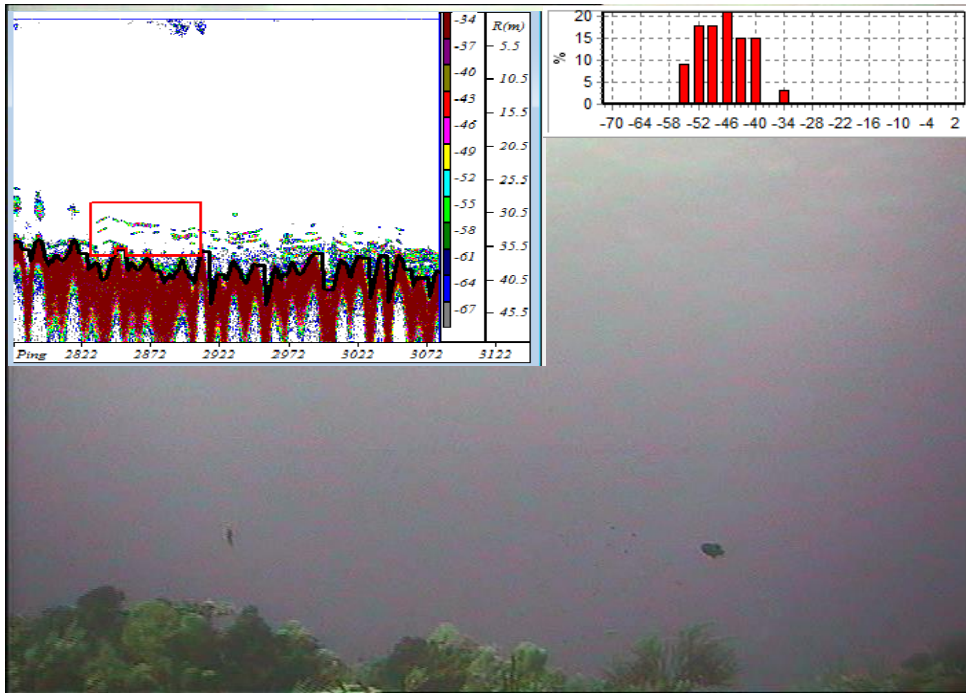
Figure 2.6 Example use of the pelagic towed camera system to determine fish species, in this case Black Durgon (Melichthys niger), with the insets showing the corresponding echogram and frequency of returning signal strengths (dB) from the fish selected by the red box.
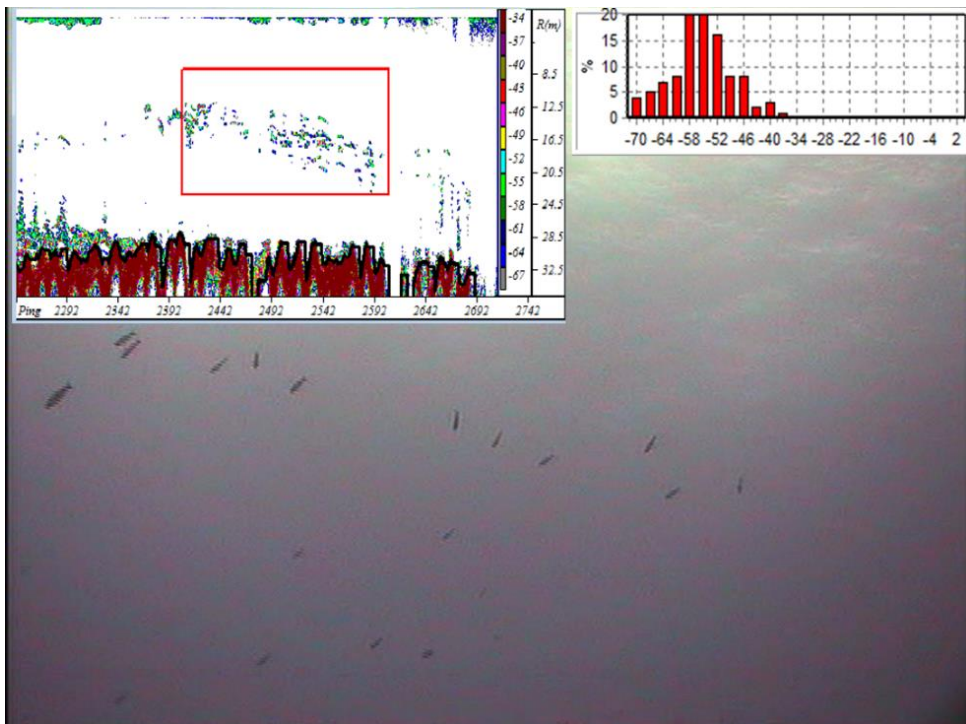
Figure 2.7 Example use of the pelagic towed camera system to determine fish species, in this case Creole Wrasse (Clepticus parrae), with the insets showing the corresponding echogram and frequency of returning signal strengths (dB) from the fish selected by the red box.

2.3 References

Balk, H. and Lindem, T. 2006. Sonar 4, Sonar 5, Sonar 6 e Post-processing Systems. Operator Manual. Oslo: Lindem Data Acquisition.

Boswell, K.M., Wilson, M.P. and Wilson, C.A., 2007. Hydroacoustics as a tool for assessing fish biomass and size distribution associated with discrete shallow water estuarine habitats in Louisiana. *Estuaries and Coasts*, *30*(4), pp.607-617.

Draštík, V., Kubečka, J., Čech, M., Frouzová, J., Říha, M., Juza, T., Tušer, M., Jarolím, O., Prchalová, M., Peterka, J. and Vašek, M., 2009. Hydroacoustic estimates of fish stocks in temperate reservoirs: day or night surveys? *Aquatic Living Resources*, *22*(1), pp.69-77.

Foote, K. G., Knudsen, H. P., Vestnes, G., MacLennan, D. N., and Simmonds. E. J. 1987. Calibration of acoustic instruments for fish density estimation: a practical guide. Cooperative Research Report. *ICES CM,, 144*, 69 pp

Mackinson, S., Freeman, S., Flatt, R. and Meadows, B., 2004. Improved acoustic surveys that save time and money: integrating fisheries and ground-discrimination acoustic technologies. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.129-140.

Simmonds, J. and MacLennan, D.N., 2005. *Fisheries acoustics: theory and practice*. John Wiley and Sons.

Winfield, I.J., Emmrich, M., Guillard, J., Mehner, T. and Rustadbakken, A., 2011. WISER Deliverable 3.4-3: Guidelines for standardisation of hydroacoustic methods. *WISER Project Newsletter*, *2*, pp.4-4.

3 Chapter 3. Hydroacoustics for the discovery and quantification of Nassau grouper (*Epinephelus striatus*) spawning aggregations

3.1 Abstract

Fish Spawning Aggregations (FSAs) are vital life-history events that need to be monitored to determine the health of aggregating populations, and this is especially true of the endangered Nassau grouper (*Epinephelus striatus*). Hydroacoustics were used to locate Nassau grouper FSAs at sites on the west end of Little Cayman (LCW), and east ends of Grand Cayman (GCE) and Cayman Brac (CBE). Fish abundance and biomass at each FSA were estimated via echo integration and assessment of FSA extent. Acoustic arithmetic mean fish abundance estimates on the FSA at LCW (893±459SE) did not differ significantly with concurrent SCUBA estimates (1150±75 SE). Mean fish densities (fish number per $1000m^3$) were significantly higher at LCW (33.13±5.62 SE) than at the other sites (GCE 7.01±2.1 SE, CBE 4.61±1.16 SE). We investigate different acoustic post-processing options to obtain Target Strength (TS) and we examine the different TS to Total Length (TL) formulas available. The SCUBA surveys also provided measures of TL through the use of laser callipers allowing an *in situ* TS to TL formula to be suggested for Nassau grouper at the LCW FSA. Application of this formula revealed mean fish TL was significantly higher at LCW (65.4 cm±0.7 SE) than GCE (60.7 cm±0.4 SE), but not CBE (61.1 cm±2.5 SE). Use of empirical TS to TL formula resulted in underestimation of fish length in comparison to diver measurements, highlighting the benefits of secondary length data and deriving specific TS to TL formula for the population to be surveyed. FSA location examined with reference to seasonal marine protected areas (Designated Grouper Spawning Areas) showed FSAs were partially outside these areas at GCE and very close to the boundary at CBE. As FSAs often occur at the limits of safe diving operations, hydroacoustic technology provides an alternative method to monitor and inform future management of aggregating fish species.

3.2   Introduction

Fish Spawning Aggregations (FSAs) are broadly defined as "a group of conspecific fish gathered for the purposes of spawning with fish densities significantly higher than are found during the non-reproductive periods" (Domeier and Colin, 1997). This reproductive strategy creates temporary concentrations of fish (Johannes, 1978; Kobara and Heyman, 2008) that are highly susceptible to overfishing (Nemeth, 2005; Starr et al. 2007; Sadovy de Mitcheson and Erisman, 2012). The health of a FSA is a good indicator for the health of the population as a whole (Gascoigne 2002) and any depletion of a FSA has serious consequences for the reproductive output of that population (Sadovy and Domeier, 2005; Sadovy de Mitcheson, 2016). FSAs therefore are important life-history phenomena that must be considered in any effort to manage fisheries of aggregating species (Sadovy and Colin, 2012; Sadovy de Mitcheson, 2016). Within this Chapter we use the term FSA for fish that are gathered together for the purpose of spawning. We acknowledge however that the aggregations of fish detected may not have been spawning *per se* at the specific times of the surveys.

One of the best known examples of the demise of a species due to FSA overfishing is that of the Nassau grouper (*Epinephelus striatus*) (Sadovy de Mitcheson et al. 2008). These large top-level predators are a highly important species within Caribbean reef ecosystems (Archer et al. 2012; Stallings, 2008, 2009). Nassau grouper are known to migrate to specific sites during periods of winter full moons to reproduce in FSAs (Sala et al. 2001; Whaylen et al. 2004; Starr et al. 2007) and were one of the first large-bodied tropical reef fish species scientifically documented to do so (Smith, 1972). It is estimated that 75% of all known Nassau grouper spawning aggregations have either been eradicated or reduced to negligible numbers (Sadovy de Micheson et al. 2008). Following over-exploitation these aggregations often fail to recover (Semmens et al. 2007; Gibson, 2007), although recent evidence suggests that effective management can lead to population increases (Heppell et al. 2012; Kadison et al. 2010). FSAs in the Cayman Islands have been reported on the Eastern and Southwest points of Grand Cayman, the Northeast and Southwest points of Little Cayman and the Southwest point of Cayman Brac (Bush et al. 2006). These sites were protected by legislation in 2003 which prohibits fishing in these areas (Whaylen et al. 2006) and due to winter spawning, it is now forbidden to take a Nassau Grouper from Cayman waters during the months of December to April (Cayman Islands Government, 2016).

3.2.1 The monitoring of spawning aggregations

Monitoring a FSA is an effective way to determine the health of an aggregating population, but adequately monitoring an FSA requires a clear understanding of its location, extent, and dynamics. In-water monitoring is fraught with difficulties including high temporal variability in fish numbers and variable distribution across multiple sites, the expense of Underwater Visual Census (UVC) surveys and challenging underwater working conditions (including strong currents, poor visibility and FSA locations below safe diver depth limits) (Sadovy de Mitcheson and Domeier, 2005). This is especially true in the Cayman Islands where FSAs occur on the extreme tips of the Islands at locations where currents are strong and dives must occur at dawn and dusk to coincide with periods of peak fish activity. Further, observer bias may be present in UVC surveys and diver avoidance by fish may occur (Colin 1992; Murphy and Jenkins 2010).

Hydroacoustics may be useful for assessing aggregating reef fishes that are otherwise difficult to count (Johannes et al. 1999). One of the main advantages of hydroacoustics is the ability to collect large volumes of information in a short amount of time (Trenkel et al. 2011; Jones et al. 2012). Further, unlike video or UVC, the acoustic technique is unaffected by underwater visibility (Gledhill et al. 1996) nor are the fish influenced by the presence of a diver. To date there has been some limited use of hydroacoustics for the monitoring of spawning aggregations (e.g. Johnston et al. 2006; Erhardt and Deleveaux, 2007; Taylor et al. 2006) and Taylor et al. (2006) noted the technology can provide an accurate estimate of overall fish abundance and spatial extent in comparison to diver visual counts. Studies comparing hydroacoustics and UVC are sparse, however. Taylor et al. (2006) reported similar acoustic density estimates to diver estimates over their entire survey region, although total abundances differed likely due to differences in area covered by the two methods and the patchy distribution of the fish. Although hydroacoustic techniques hold great promise, many authors highlight that ground-truthing is required to identify the fish to species level (Simmonds and MacLennan, 2005, Ryan et al. 2009).

The International Union for the Conservation of Nature (IUCN) lists the Nassau grouper as Endangered and recommends annual monitoring of as many traditional aggregation sites as possible, including adjacent areas where aggregations have not previously been reported and as part of the assessment of the effectiveness of protected areas (Carpenter et al. 2015). Given the need to develop effective monitoring techniques that can rapidly, effectively, and

quantitatively assess FSA status, we investigated the capacity of hydroacoustics to address these recommendations.

In this chapter acoustic fish abundance data were compared with diver-collected data (Thesis Hypothesis 1). FSA locations were also examined in relation to protected zones in the Cayman Islands (Thesis Hypothesis 2). Further, the different acoustic processing methods available to estimate the sizes of fish within FSAs were evaluated.

3.3 Methods

3.3.1 The Survey Sites

The sites chosen in this study are all within the Designated Grouper Spawning Areas (DGSA) of the Cayman Islands. Surveys were focussed on the likely areas of the FSAs, based on site geomorphology and from local knowledge via the Department of Environment (DoE) (Figure 3.1). Most survey effort was concentrated on the FSA located at the west end of Little Cayman (LCW) as this is known to be the most active of the FSAs, and for which concurrent fish abundance and size data obtained via SCUBA was provided by the Grouper Moon Project (http://www.reef.org/groupermoonproject). Surveys were also conducted at Little Cayman East (LCE), Grand Cayman East (GCE) and Cayman Brac West (CBW) and East (CBE). The field surveys in Cayman occurred between the 14th and 20th of February 2014 (see

Table 3.1 for details).
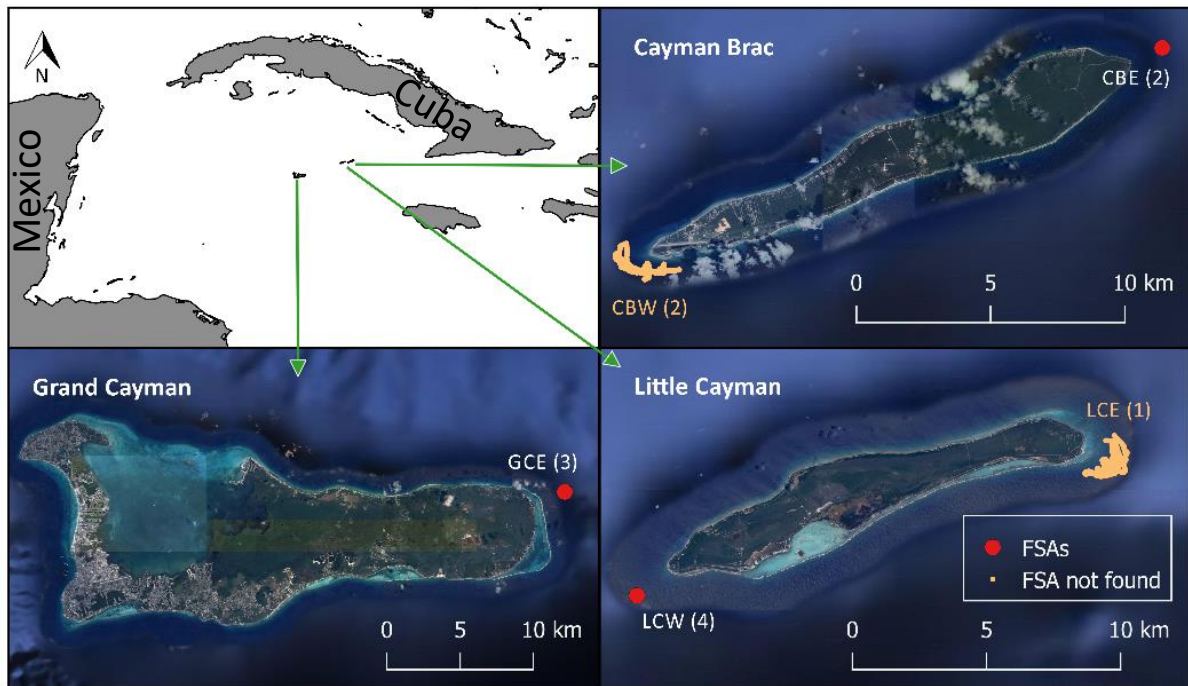
Figure 3.1. Overview map of areas surveyed by hydroacoustics and in-water assessment techniques (Map data ©2016 Google). The numbers at each site represent the total number of hydroacoustic surveys undertaken at each location. Red dots show located FSAs whereas peach colour shows survey track which did not locate FSAs. LCW: Little Cayman West, LCE: Little Cayman East, GCE: Grand Cayman East, CBW: Cayman Brac West, CBE: Cayman Brac East. Inset map shows the broader location of the Cayman Islands.

Table 3.1 The dates and times of the surveys conducted, with the number of days elapsed since the February full moon. LCW: Little Cayman West, LCE: Little Cayman East, GCE: Grand Cayman East, CBW: Cayman Brac West, CBE: Cayman Brac East.

Survey name	Date	Start time	Stop time	Days after Full Moon
GCE1	14/02/2014	12:40:43	15:19:39	0
LCE1	15/02/2014	17:48:01	19:33:54	1
LCW1	16/02/2014	12:04:38	12:52:39	2
LCW2	16/02/2014	17:38:42	17:51:19	2
LCW3	16/02/2014	18:38:18	19:12:52	2
LCW4	17/02/2014	13:24:40	13:55:05	3
CBW	17/02/2014	17:05:56	18:25:45	3
CBE	18/02/2014	17:44:52	19:00:25	4
CBW2	18/02/2014	10:43:05	13:04:03	4
CBE2	19/02/2014	07:43:09	08:48:04	5
GCE2	19/02/2014	17:13:11	18:28:32	5
GCE3	20/02/2014	08:13:58	09:41:08	6

Times are in Eastern Standard Time (EST) (UTC/GMT -5 hours)

3.3.2 Equipment

A Biosonics® DTX Split-beam echosounder with a 200 kHz transducer (beam opening angle of 6.8°), pole mounted over the side of the survey vessel was used for the surveys with the face 0.5 m under the water surface. Data were collected with Biosonics Visual Acquisition 6 software (Biosonics 2010). Pulse duration was 0.4 ms and the specified ping rate was 10 per second. Survey speed was kept to approximately 4 knots (2 m/s) and sea state was calm (Beaufort scale 3 or under) on all surveys. The echosounder was calibrated before the start of the surveys on 13/02/2014 using a tungsten carbide 36 mm standard calibration sphere, following the standard methods (Foote et al. 1987, Demer et al. 2015). The acoustic return from the sphere was within acceptable tolerance to the expected value given for the local environmental settings (TS = -39.6 dB vs -39.8 dB, respectively (Biosonics, 2004), with speed of sound calculated as 1521.54 ms^{-1}) with seasurface temperatures provided by DoE staff. Where diver observations were not available for species groundtruthing, underwater video was

used (Thomas and Thorne, 2003; Doray et al. 2007; Jones et al. 2012). This consisted of a Sony 37CSHR camera, with a live surface feed, mounted on an aluminium wing. Both the acoustic data and the video data were time-stamped allowing synchronisation of the visual and acoustic records in post-processing.

3.3.3 Data processing methods

Potential Nassau grouper FSAs were initially identified through their stronger backscattering properties and school morphology (Figure 3.2) than aggregations of other species (e.g. Horse-eye Jack *Caranx latus*), and then verified by visual observation either by the use of the pelagic tow camera or through confirmation by the dive team at LCW.
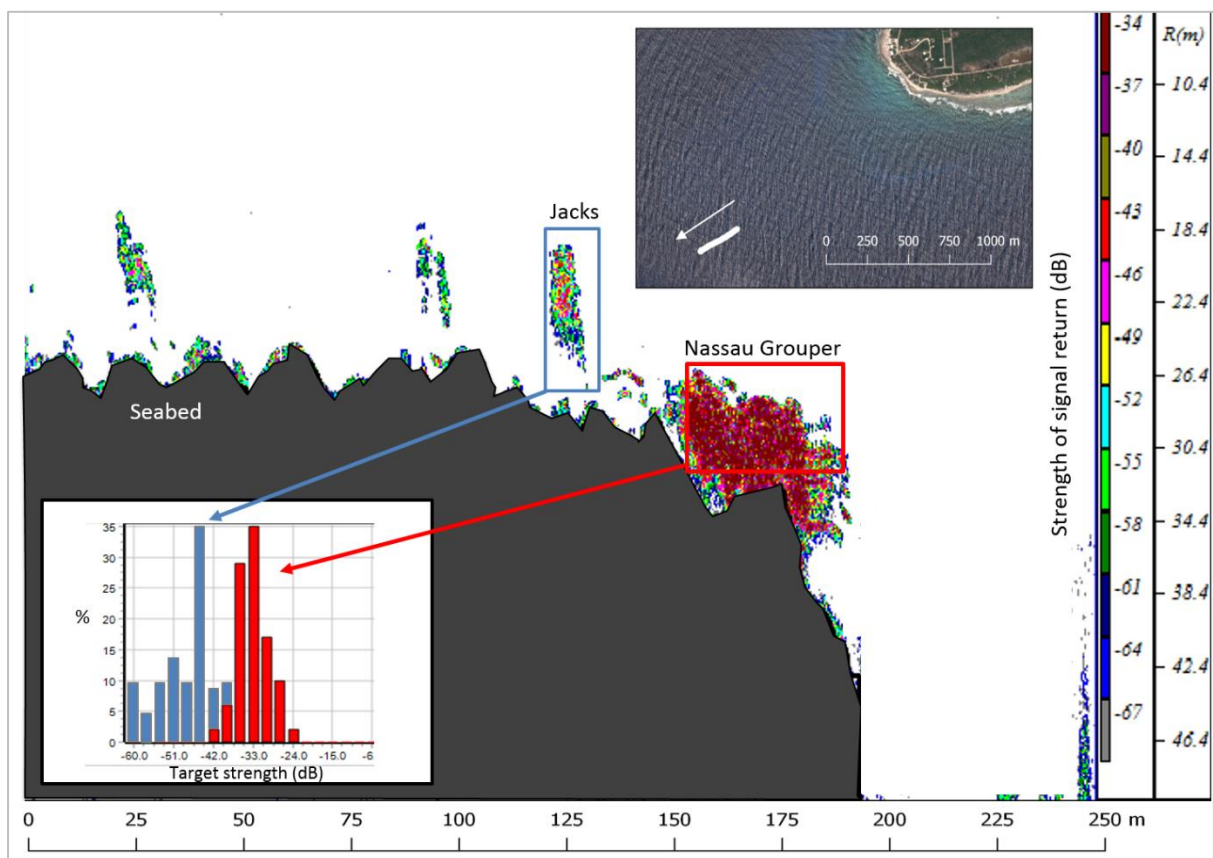
Figure 3.2 An example echogram of the analysis of fish echoes resulting from a Nassau Grouper fish spawning aggregation (FSA) (red) and those from an aggregation of Horse-eye Jacks (blue). The resultant signal strength graph (inset) shows the Grouper as having a higher percentage of stronger echoes. Transect distance is shown along the x axis while depth (R(m)) and strength of signal return (colour strip) is shown on the y axis. The inset satellite image (Map data ©2016 Google) shows the location of the transect conducted on the first Little Cayman West (LCW)1 Survey and the arrow shows the direction of travel.

Data were processed with the software package Sonar5 (ver 6.0.4)-Pro (Balk and Lindem 2006), following the Software Guided Analysis (SGA) routine (see Parker-Stetter et al. 2009

for details). The analysis was based upon echo integration (also known as Sv/TS scaling) which divides the average reflection from all fish over a segment (the volume backscattering coefficient, Sv) by the average Target Strength (TS) from individual fish (Winfield et al. 2011). TS is defined as: **TS = mLogL +b** where m and b are constants for a given species and frequency, respectively (Simmonds and MacLennan, 2005). Initially a threshold of -60 dB was applied to the echograms to distinguish fishes from other particulate targets such as plankton, this is a typical threshold applied for the detection of pelagic schooling fishes (ICES, 2000). Any noise due to bubbles in the water column from wave action was removed by eye. Sonar5 (ver 6.0.4) applies a Time Varied Gain (TVG) correction of 40log(R) for TS values and 20log(R) for Sv values (Balk and Lindem, 2006). A bottom exclusion layer of 1 m was applied and data from within this layer were not included in the analysis due to the 'acoustic dead zone' (Ona and Mitson, 1996). For echo integration methodology there are two main options to obtain TS; using tracked fish as a source or using 'single echoes detected' (SED) as source. We used tracked fish as source in order to derive abundance estimates but examined the efficacy of both options to derive TS. In order to track fish within the FSAs we used the following default Sonar5 (ver 6.0.4) criteria of; a minimum track length of 3 pings, with a maximum ping gap of 2 pings, a gating range of 0.3 m and applied a maximum mean echo threshold of -25 dB and a minimum of -40 dB. Due to difficulties in obtaining sufficient numbers of tracks from within FSAs (likely due to high fish density and low signal to noise ratios in dense areas of the aggregation), tracks were extracted and stored from all passes of the FSAs per survey and then these tracked fish were used to provide the survey-specific abundance estimates. As tilt angle of fish can have a significant bearing on TS (and subsequent abundance and density estimates), extreme tilt angles were filtered out of the data following Gauthier and Horne (2004), so that any fish with an aspect ±40deg from horizontal (dorsal aspect) were removed from the analysis. We examined both the mean TS of fish echoes in each track (calculated in the linear domain) and the 75th percentile of TSs of each track. For fish TS estimates using SED as source, SED were extracted for each pass of an FSA and mean TS values subsequently determined for the FSA from each survey. In order to assess that fish near the top of a school were not shadowing those beneath them, data were checked to ensure that echo energy was consistent from the top to the bottom of the school following Knudsen et al (2009) (See Appendix 1 Figure 3.8).

In order to convert TS to fish Total Length (TL) three main equations were examined by applying our mean TSs values (Table 3.2). Further, we scaled diver fish length (TL)

measurements (taken using a laser calliper system; Heppell et al. 2012) by our mean TS data from tracked fish for the LCW FSA, by sorting both datasets by increasing value and then plotting one against the other to determine a survey-specific TS-TL formula (see Appendices Figure 3.9) resulting in Formula 4 in Table 3.2.

Table 3.2 The target strength (TS) to total length (TL) formulae examined in this study.

	Formula TS to L	Formula L to TS	Reference	Species	Frequency
1	TS = 19.1 log10(L) - 64.07	L=(2261.8)*EXP(0.1206*(TS))	(Love 1971)	Multi species	200 kHz
2	TS = 0.7091*L - 89.136	L= (TS/0.7091)+89.136	(Erhardt and Deleveaux, 2007)	*Epinephelus striatus*	200 kHz
3a	TS = 19.2 log10(L)- 64.05	L=(2165)*EXP(0.12*(TS))	(Rivera et al. 2010)	*Epinephelus guttatus*	120 kHz
3b*	TS = 19.2 log10(L)- 64.25	L=(2220)*EXP(0.1199*(TS))	(Rivera et al. 2010)	*Epinephelus guttatus*	200 kHz
4	TS = 27.6 log10(L)- 147.32	L=(207.06)*EXP(0.0362*(TS))	This manuscript	*Epinephelus striatus*	200 kHz

*3b is 3a reformulated for 200 kHz

TL - weight regressions specific to the Nassau grouper were used to calculate weight at TL for biomass estimates using the formula: $\mathbf{W = a\,L^b}$ where W = weight (g), L= TL (cm), a = 0.01122, b = 3.05 (Froese et al. 2016).

Applying the TS- TL formula and then using the specific TL to weight relationship for the Nassau grouper (Froese et al. 2016) gives the mean weight of fish in each FSA. This figure was then multiplied by the number of fish estimated in each FSA to provide total biomass estimates for each FSA surveyed.

3.3.4 Spatial extents

Once the FSA was located using preliminary acoustic transects, the aggregation was surveyed from different angles to corroborate its extent. This approach follows Doonan et al. (2003), who note the advantages of a star-shaped survey track in hydroacoustic surveys over schooling fishes. Alongside fish abundance values, the geographical extents were also extracted; however these are given only in two dimensions (height and length). Where survey tracks crossed the FSA from different angles, the full 3 dimensional extent of the FSA was estimated by drawing a polygon (Figure 3.3) as per the arithmetic extrapolation method used by Taylor et al. (2006)

and Erhardt and Deleveaux (2007). When the track crossed the FSA from one angle only, it was assumed that the aggregation was circular in shape unless nearby pings showed no fish were present in which case the half way point between the positive (FSA detected) and negative (FSA not detected) pings was taken to demarcate the FSA extent. If the FSA represented two or more clear densities, then separate polygons were drawn for each density class present. Once a polygon was drawn, fish abundance was calculated by multiplying the mean number of fish per ha by the area of the polygon. When there were multiple polygons of differing abundances then the results of each was summed to give a total number of fish.
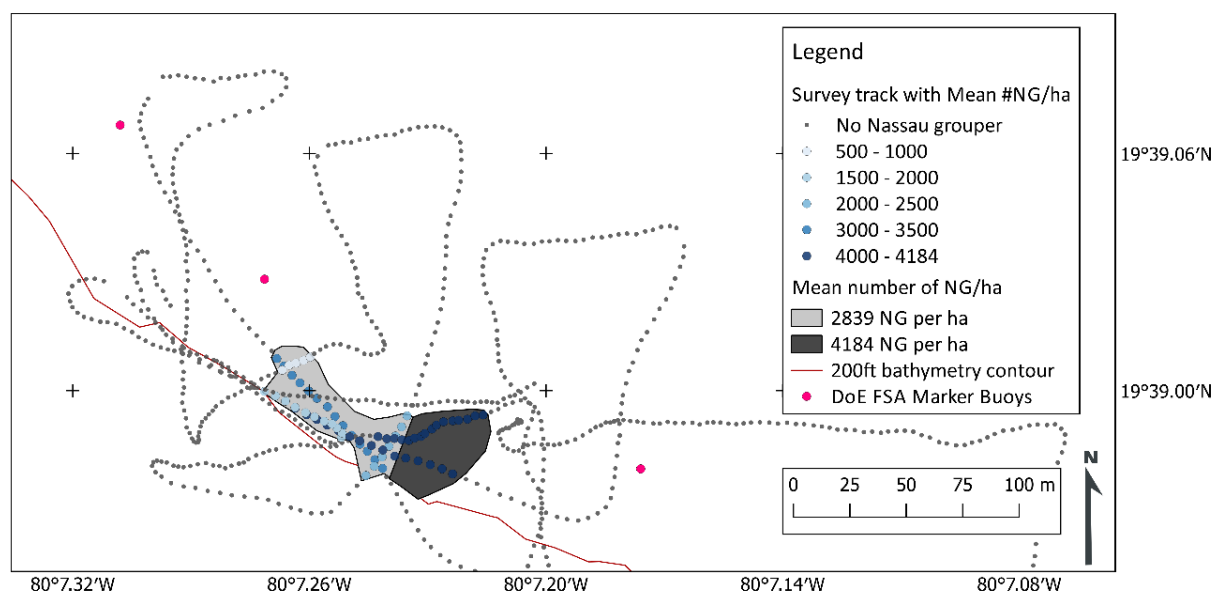
Figure 3.3 Example of FSA polygon determination in the Arithmetic extrapolation method during the LCW4 survey. NG = Nassau grouper. Also shown are DoE's Little Cayman FSA location marker buoys and the 200ft (61m) bathymetry contour.

3.3.5 Statistical Analyses

Welch's ANOVAs (equal variances were not assumed) were used to compare fish densities (number of fish/1000 m^3) (log transformed) between sites and between surveys at LCW, whilst a 2 sample t-test was used to test this at GCE surveys. Diver fish abundance estimates were compared to the acoustic abundance estimates by using a two sample t-test. The TS values resultant from the different acoustic processing methods were compared for each site with 2 sample t-tests. Values of fish TL gained from applying tracked fish mean TS data coupled with our *in situ* formula were compared between the different surveys and sites also with Welch's

48

ANOVA and Games-Howell Pairwise Comparisons were used to test where the differences between sites existed.

3.4 Results

3.4.1 Numbers of fish in each FSA

FSAs were identified at LCW (all 4 surveys), GCE (2 of 3 surveys) and CBE (1 of 2 surveys). No FSAs were detected on the surveys of CBW nor LCE. Visual confirmation that the targets were Nassau grouper was provided by the Grouper Moon dive team at LCW, and at GCE by the towed camera system. We did not achieve visual confirmation of species present at CBE; however mean TS's and FSA morphology at that location were similar to those at the verified Nassau grouper FSA sites. The highest acoustically measured fish abundance was detected at LCW with a maximum abundance of 2194 fish in the aggregation (survey LCW1) 2 days after the full moon on 16/02/2014. Fish density was significantly higher at LCW FSA than at the other two sites ($F_{1,2}=25.49$, $p<0.001$) which did not differ significantly from each other. Fish densities did not differ significantly between individual surveys at the LCW FSA ($F_{1,3}=1.35$, $p=0.32$) or the GCE FSA ($T_8=1$, $p=0.35$) (Table 3.3).

Table 3.3 Estimates of mean TS, mean lengths, weights, fish numbers, and subsequent biomass values per survey where a FSA was identified as derived from mean TS from tracked fish in the Cayman Islands. LCW: Little Cayman West, LCE: Little Cayman East, GCE: Grand Cayman East, CBW: Cayman Brac West, CBE: Cayman Brac East.. Fish density is number of fish per 1000m³. Nv is number of fish per volume isonified (Sawada et al. 1993). Verification method shows how the fish were identified "D" = Diver (number in brackets), "NP"= Not Possible, "TC" = Towed Camera. Mean depth is the mean fish depth at each FSA. Numbers in brackets are 95% confidence levels.

Survey name	Mean TS (dB)	Mean Length (cm)	Mean Weight (g)	Fish number	Biomass (kg)	Verification method (diver fish number)	Fish density (#/1000m^3)	Fish/ isonified volume (Nv)	Mean Depth (m)
LCW1	-31.98 (0.86)	65.22 (2.06)	3900.03 (390.9)	2194	8556.67	D (NP)	46.89 (24.60)	0.095 (0.05)	28.0 (1.4)
LCW2	-32.89 (1.43)	63.60 (3.30)	3782.35 (598.3)	398	1505.37	D (1225)	24.69 (12.76)	0.051 (0.024)	28.9 (2.1)
LCW3	-32.62 (1.25)	63.94 (2.97)	3746.54 (559.0)	122	457.08	D (1225)	18.20 (5.29)	0.031 (0.007)	26.2 (2.6)
LCW4	-30.50 (0.84)	68.86 (2.11)	4615.64 (443.8)	857	3955.60	D (1000)	32.87 (21.50)	0.072 (0.046)	29.0 (2.6)
LCW all	-32.01 (0.61)	65.40 (1.44)	4018.20 (268.1)	893	3588.25	D	33.13 (11.02)	0.067 (0.023)	28.1 (1.1)
CBE1	-33.95 (2.26)	61.12 (5.08)	3327.10 (849.2)	58	192.97	NP	4.61 (2.27)	0.009 (0.005)	30.4 (1.9)
GCE2	-33.95 (0.55)	60.90 (1.2)	3208.22 (191.6)	49	157.20	TC	4.01 (2.24)	0.0198 (0.011)	43.7 (2.2)
GCE3	-34.07 (0.48)	60.61 (1.08)	3162.43 (181.7)	40	126.50	TC	8.37 (5.82)	0.042 (0.028)	46.1 (1.1)
GCE all	-34.01 (0.36)	60.74 (0.8)	3183.32 (131.6)	45	143.25	TC	7.01 (4.12)	0.035 (0.019)	45.2 (1.1)

3.4.2 Comparison between acoustic and diver abundance data

Diver-estimated numbers of fish at the LCW FSA were made concurrent with acoustic surveys LCW2, LCW3 and LCW4 (Table 3.3). Diver confirmation of species also occurred during LCW1, although numbers could not be recorded. No significant difference was detected at the 95% confidence level between diver estimates and acoustics (T_3=0.55, p=0.62).

3.4.3 Fish TS

Mean fish TS gained through tracked fish was compared with mean fish TS via SED for each site. This analysis revealed that there was no significant difference in resultant mean TS values at any site: (CBE: T_{12}=0.03, p=0.98, LCW: T_{47}=1.44, p=0.157, GCE: T_{28}=0.59, p=0.56). The TS values from the 75th percentile of echoes in a fish track was significantly higher than the mean TS at LCW (T_{192}=3.78, p>0.001) and GCE (T_{429}=6.91, p>0.001), but not at CBE (T_{19}=1.13, p=0.273) presumably due to the lower number of observations reducing statistical power. The data on TS from applying the different methods is shown in Figure 3.4.
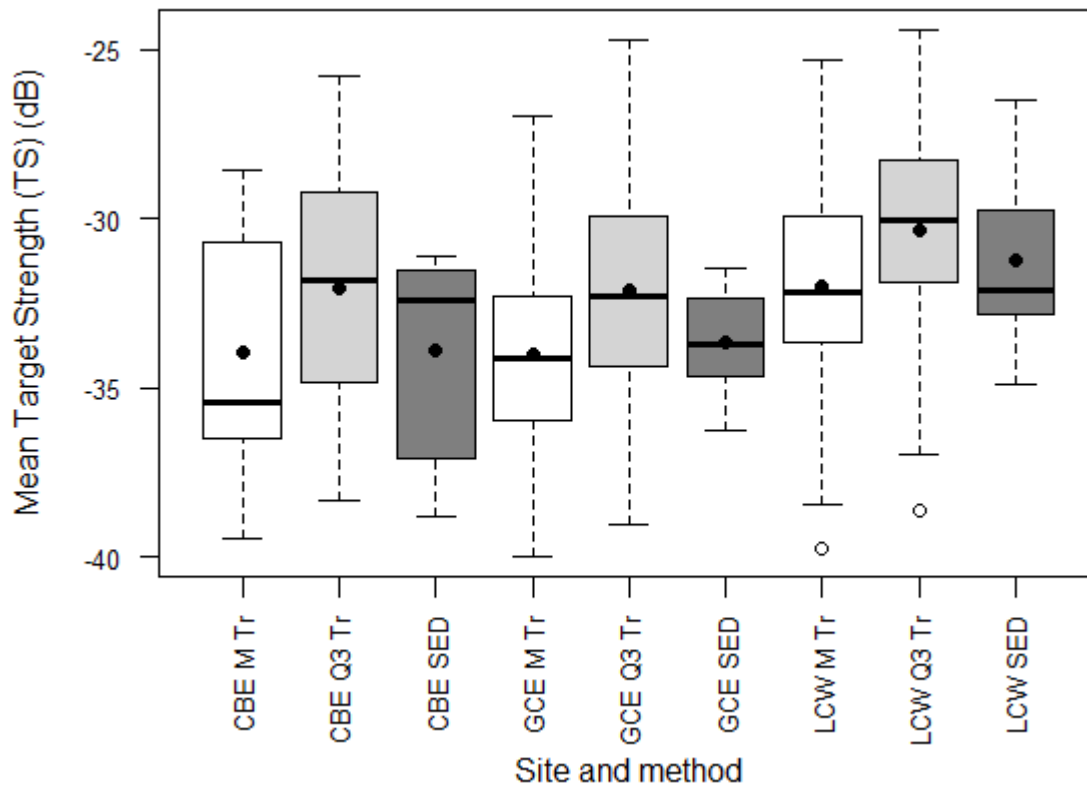
Figure 3.4 Mean fish target strength (TS) found in FSAs during each survey, per site and for each of the acoustic processing methods. Tr M is the mean echo of tracked fish, Q3 Tr is the 75th percentile of echoes from tracked fish, SED is Single Echoes Detected. Box plots show mean values (black circle), median values (solid horizontal line), and the lower and upper ends of the box are the 25% and 75% quartiles respectively. The whiskers indicate 1.5 times the inter-quartile range and points beyond this range are shown by empty circles.

3.4.4 Converting TS to TL

Mean TS measurements from tracked fish were scaled by the diver LCW FSA diver length data. This resulted in: **TS= 27.6log10(L)-147.32** (R^2 of 0.98) (see Appendix 2 Figure 3.9). The results of then applying this formula to TS data are plotted for the LCW dataset below alongside the alternative equations given in Table 2 (Figure 3.5, opposite).
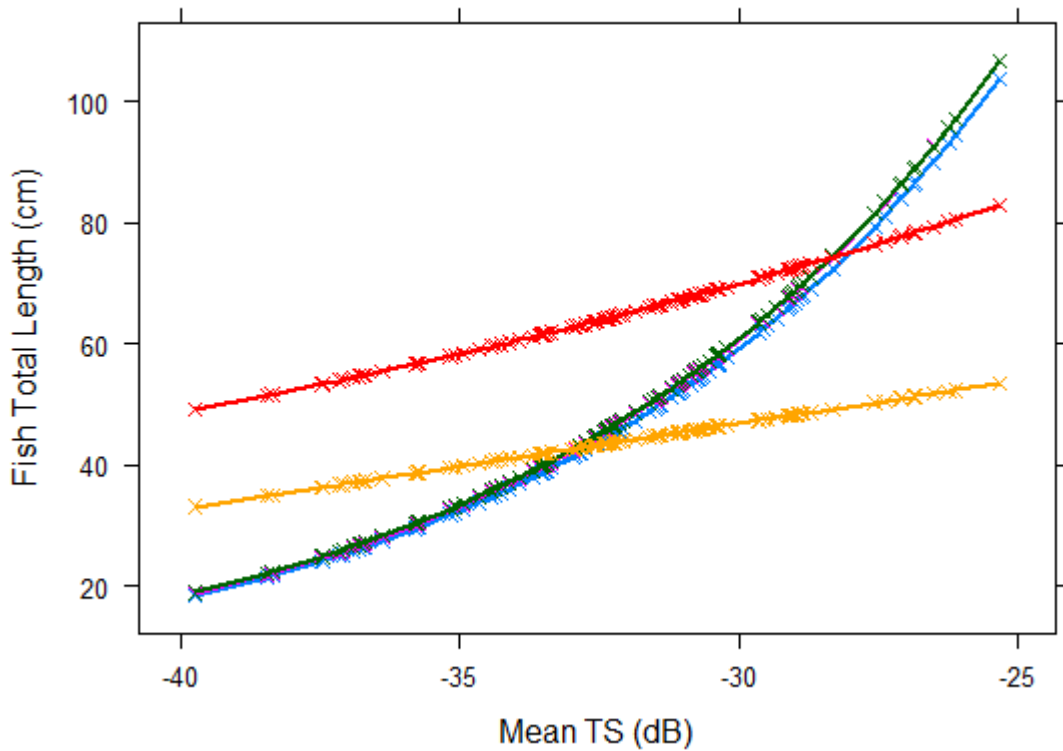
Figure 3.5 Target Strength (TS) data from the LCW surveys and corresponding fish total length using the following empirical formulas: Rivera et al. 2010: TS = 19.2 log10(L)-64.05 (Blue) and reformulated for 200khz (Green), Love 1971: TS = 19.1 log10(L)-64.07 (Pink, partially hidden due to similar values as green), Erhardt and Deleveaux, 2007: TS = 0.7091*L -89.136 (Yellow), Our formula: TS = 27.6 log10(L)-147.32 (Red).

The results of applying our *in situ* formula to the acoustic TS data are plotted per individual survey (Figure 3.6a) and as mean values per site (Figure 3.6b).
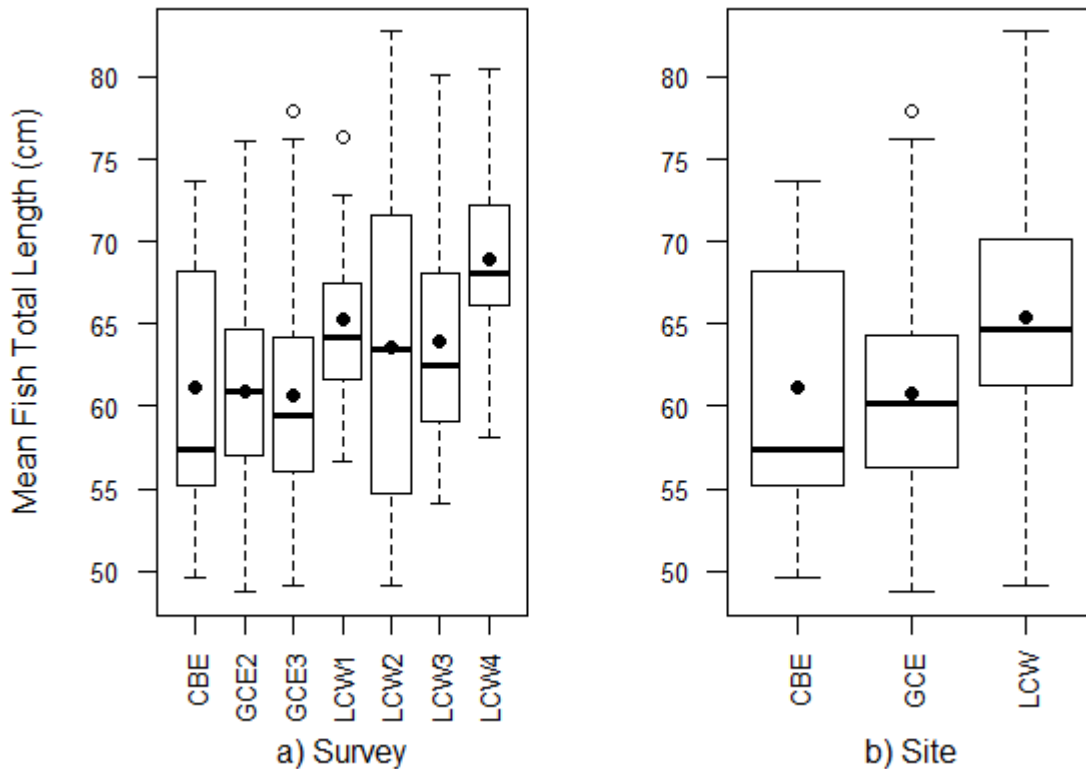
Figure 3.6 Mean fish total length (TL) as calculated by applying our in situ formula (a) during each survey, and (b) as grouped data per site. Box plots show median values (solid horizontal line), and the lower and upper ends of the box are the 25% and 75% quartiles respectively. The whiskers indicate 1.5 times the inter-quartile range and points beyond this range are shown by empty circles.

There was a significant difference in mean fish TL calculated from mean TS of tracked fish between the sites (F_2,=15.08, p>0.001), with significantly larger fish at LCW than at GCE but not CBE, which did not differ from each other. Using the von Bertalanffy growth curve for the Nassau grouper sampled from aggregations in the Cayman Islands 1987 – 1992 (Bush et al. 2006), the estimated mean fish TL of 65.4 (±0.7SE) seen at the LCW FSA corresponds to an age of 10 years. The estimated mean sizes of fish at the GCE FSA (60.7 cm ±0.4SE), and CBE (61.1 cm ±2.5SE) correspond to that of 8 year old fish.

3.4.5 FSA location relative to Cayman Islands Department of Environment Designated Grouper Spawning Areas (DGSA)

The extent of the FSA located on Grand Cayman fell on the extreme northern limit of the DGSA boundary on the GCE2 survey and just outside the boundary during the GCE3 survey.

At CBE the FSA was just within the boundary close to its northern limit. The LCW FSA was within the associated protection zone (Figure 3.7).
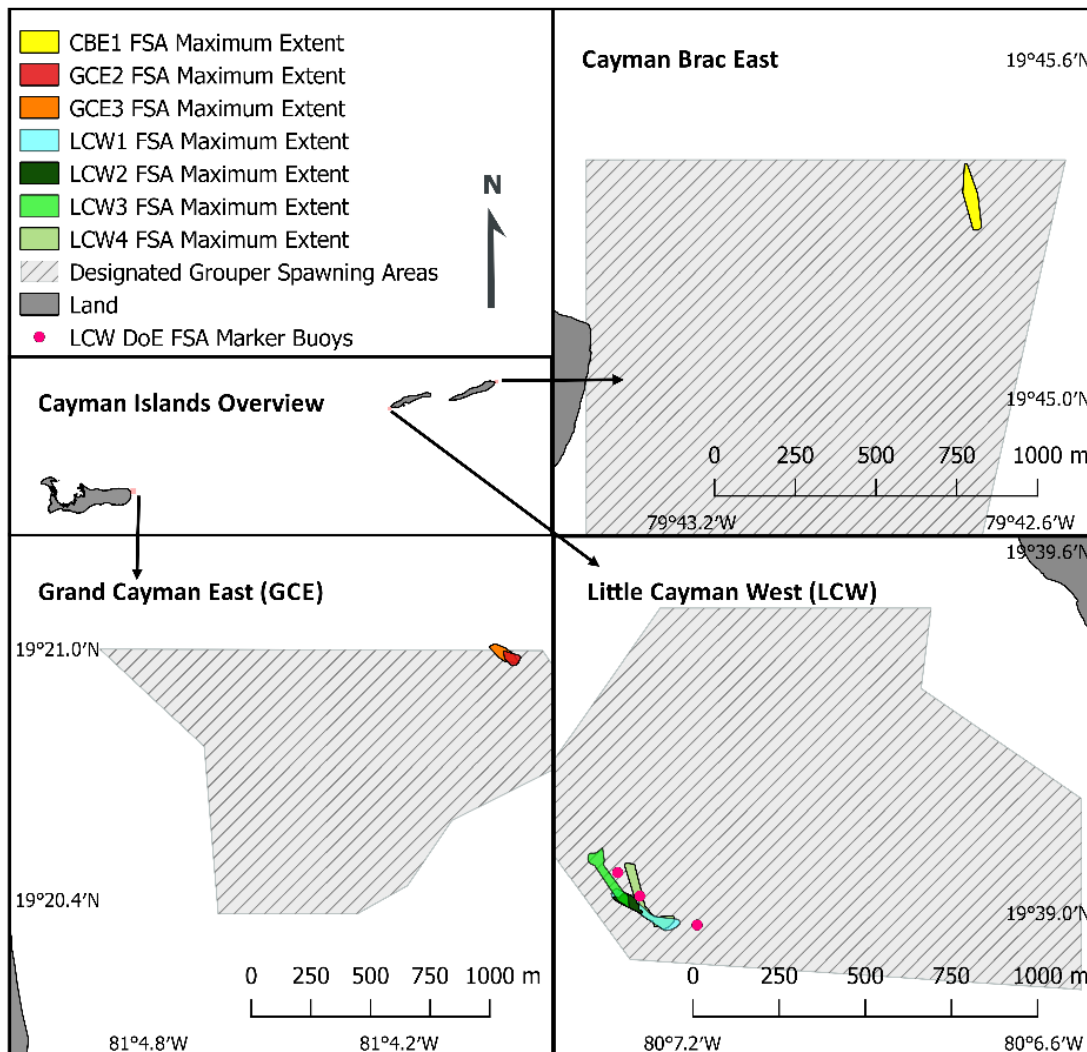
Figure 3.7 FSA locations and maximum extents detected via hydroacoustics in the Cayman Islands in relation to the positions of the Designated Grouper Spawning Areas (DGSAs).

3.5 Discussion

The highest fish abundances and densities were recorded at the LCW FSA. This is as expected as this particular FSA is well known throughout the Caribbean for the high numbers of fish present there during spawning periods (Whaylen et al. 2004). It should be noted that these surveys occurred closest to the full moon (2-3 days after the full moon), when Nassau grouper FSAs are most active (Starr et al. 2007). The surveys LCW1 and LCW4 both yielded very similar patterns of fish distribution and had the highest abundance estimates. These surveys occurred at similar times near the middle of the day, whilst surveys LCW2 and LCW3, both occurring near dusk, recorded lower abundances. Other studies have found groupers to be more densely aggregated during sunrise and sunset (Whaylen et al. 2006), and it is possible that the main aggregation may therefore have been missed by surveys LCW2 and LCW3, or that abundance estimates are more robust when fish are more dispersed as has been seen in other studies (Rudstam et al. 2003).

At any given point in time on the LCW FSA, some proportion of the fish are located on the plateau and across a wider area than is represented by the main aggregation at the reef crest (Whaylen et al. 2006); it is possible that the acoustics may not have detected these individuals. In addition, as fish within 1 m of the seabed were not included in the study, acoustic abundance estimates are best considered an index of abundance rather than an absolute abundance and are likely to be conservative compared to the totality of all spawning fish. The LCW FSA was most active the day before the acoustic surveys (15th February, 1 day after the full moon) with 4000 fish estimated by the dive team. Our peak number of fish was detected the following day. The CBE FSA was surveyed 4 days after the full moon and the FSA at GCE surveyed 5 and 6 days after the full moon; only small numbers of fish were found at either location. It is likely that the acoustics results presented herein underestimate the total abundances of individuals in these FSAs as they do not account for the most active times i.e. closer to the full moon. Therefore we recommend that in order to fully evaluate a given FSA, acoustic surveys should be conducted both across days and at multiple times per day in order to increase the probability of capturing peak abundance at any given FSA. It should be noted that we assumed that all echoes from within a FSA were Nassau grouper but it is possible that relatively low numbers of other fish species were also present.

We evaluated the possibility of acoustic shadowing leading to the differences between diver estimates and acoustic estimates of fish numbers. No decrease in echo energy from the top of

the FSAs to the bottom was found, indicating that the acoustic technique can be used to accurately quantify fish in FSAs (Knudsen et al. 2009). This is contrary to some other studies however which have reported a shadowing effect in dense schools of marine fishes (Zhao and Ona, 2003; Utne and Ona, 2006; Løland et al. 2007).

We examined three different methods in the acoustic post processing to extract TS values, and it is interesting to note that mean TS with SED as source did not differ significantly to the mean TS of tracked fish. When fish are tilted further from the horizontal, TS is reduced so max TS may be a better estimator than mean TS (Balk Lindem, 2006). However to remove any effect of 'flash echoes' (Lilja et al. 2004) and also the potential exaggerating effects on mean TS of multiple echoes (Soule et al. 1995, Rudstam et al. 2003), a 75th percentile of the TS along a tracked fish was also examined and unsurprisingly yielded overall higher values than the other two methods. We however used the mean TS for subsequent calculations as this method is most common in the literature (e.g. Rose, 2009; Guillard et al. 2004).

TS varies both with tilt angle (Nielsen and Lundgren, 1999), and among fish species due to anatomical differences in the size of the swim bladder (Simmonds and MacLennan, 2005). As such, an empirical TS- TL relationship is needed to convert TS to fish TL, which are known for many species (Kracker 2007). Ideally, TS data should be obtained from fish that are typical of the population to be surveyed (Simmonds and MacLennan, 2005). The LCW FSA presented a rare opportunity to do this as the fish species (almost entirely Nassau grouper) could be determined by divers who were also able to provide accurate length measurements. By scaling our TS values by the diver measurements we derived an alternative *in situ* TS-TL equation allowing comparison to the other equations examined. Application of either the Love (1971) or Rivera et al. (2010) formula results in a significant underestimation of fish size in comparison to the diver data. Although our equation contains a log function it is more similar to the Erhardt and Deleveaux (2007) than the other equations. This is likely to be due to the relatively narrow range of fish sizes in both their and our studies, as these are the lengths of reproductively active fish. Whilst applying our equation matches diver lengths at LCW we are hesitant to suggest without further evaluation that it should be used in preference to other equations in future studies due to a number of reasons: There was a relatively narrow range of fish lengths present in the FSA as seen by divers, and applying our formula may have the effect of over estimating the size of smaller fish and under estimating the size of larger fish beyond the range of what was experienced here. Secondly there are difficulties in extracting tracked fish TS data from the centre of FSAs and it may be the case that the tracked fish, more

commonly located on the periphery of the aggregation, maybe of a different size or orientation than those in the centre (Starr et al. 2007). Thirdly, tracking fish is difficult in vertical marine applications (Guillard et al. 2004) and although we experienced calm sea states, vessel movement is likely to have reduced the number of possible tracks and increased variation in TS. We recommend further examination of the TS – TL relationship for Nassau grouper and caged fish experiments, or similar, should be conducted across a larger range of fish sizes to gain more empirical data points from which a potentially more robust equation can be determined. Future research examining the novel combination of using hydroacoustics and laser callipers would prove useful for FSA monitoring and other assessments of fish populations. The effect of reproductive state on TS of Nassau grouper would also be worthy of examination, as it has been seen in spawning sardines that the relationship of gonad size to swimbladder volume is as important as the relationship of the swimbladder volume to fish length (Machias and Tsimenidis, 1995). The analysis of different sizes of fish at the different FSA sites showed that the mean fish TL was significantly larger at LCW than at GCE, but not CBE. As younger fish tend to be smaller, a recovering population may have a larger proportion of smaller fish (Heppell et al. 2012). Our results could indicate that the FSAs on GCE and CBE may be recovering from previous exploitation (Bush et al. 2006) or that the generally smaller fish at those locations are a result of larger fish being removed by fishing.

Hydroacoustics allowed us to determine the location of FSAs in 3-dimensional space. Spawning aggregations were consistently found just off the reef crest at around 30 m depth at LCW as has been described previously by direct observation (Whaylen et al. 2004). The depths of FSAs will be influenced by a number of factors such as diurnal time of survey or lunar phase (Starr et al. 2007), however knowing the depths from our surveys may assist managers in determining optimum future survey strategies. The relatively deep FSA of GCE was also noted by Kobara and Heyman (2008) and is most likely to be due to the spawning suitability of the local geomorphologic characteristics at the site. The depth at which this FSA occurs highlights the difficulty of visual census approaches using SCUBA. FSAs can move between repeat surveys within the same lunar period, and some wider movement not detected in this study could reasonably be expected. Through examination of the FSAs relative to the DGSAs we managed to determine how well the FSAs were afforded protection (Thesis Hypothesis 2b). We recommend considering line fishing to be also included in the one mile radius restrictive buffers around the DGSAs or increasing the size of the DGSAs as a further precautionary measure. If fishing occurs at the edge of the protected areas, as is common practice following

closures to fishing (Kellner et al. 2007), then it is possible that these FSAs, which may be recovering, could still be at risk.

Hydroacoustics has proven to be capable of locating FSAs in historic areas where it was unknown whether fish were still aggregating (Thesis Hypothesis 2a). This also means that acoustics can be used to search for aggregations in new locations and used in situations when diving surveys are impractical or hazardous. We have shown that surveying FSAs with hydroacoustics produces fish count information comparable to that from diver estimates, confirming Thesis hypothesis 1. Furthermore, hydroacoustics provides additional information such as fish size when groundtruthing is also provided, although further work is needed in this area. Repeating hydroacoustics surveys could yield much information on how exploited FSAs are recovering and could assist with the vital monitoring of endangered aggregating populations.

3.6 Appendices

3.6.1 Appendix 1

In order to assess that fish near the top of a school were not shadowing those beneath them, data were checked to ensure that echo energy was consistent from the top to the bottom of the school following Knudsen et al. (2009)
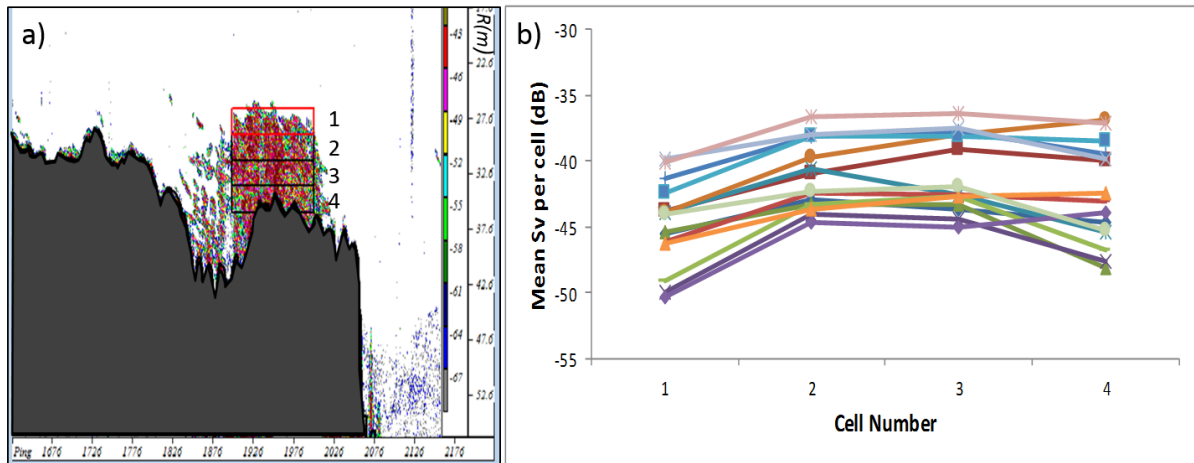
Figure 3.8 Shadow effect analysis of echoes from within an FSA. a) Four vertical analysis cells were defined in each FSA (indicated by the stack of rectangles). The x axis shows depth (R(m)) and strength of signal return (dB), ping number is shown on the y axis. b) Mean Sv was calculated for each cell and compared for 15 separate acoustic detections of FSAs along the survey track from the top (1) to the bottom (4) of the FSAs. The lack of attenuation of echoes from deeper cells indicates absence of acoustic shadowing.

3.6.2 Appendix 2

At the LCW FSA fish length data were provided by divers. We then scaled our mean TS measurements from tracked fish by these values, which resulted in the relationship given in the Figure 3.9 below. As can be seen this resulted in a strong positive relationship with an R^2 value of 0.98.
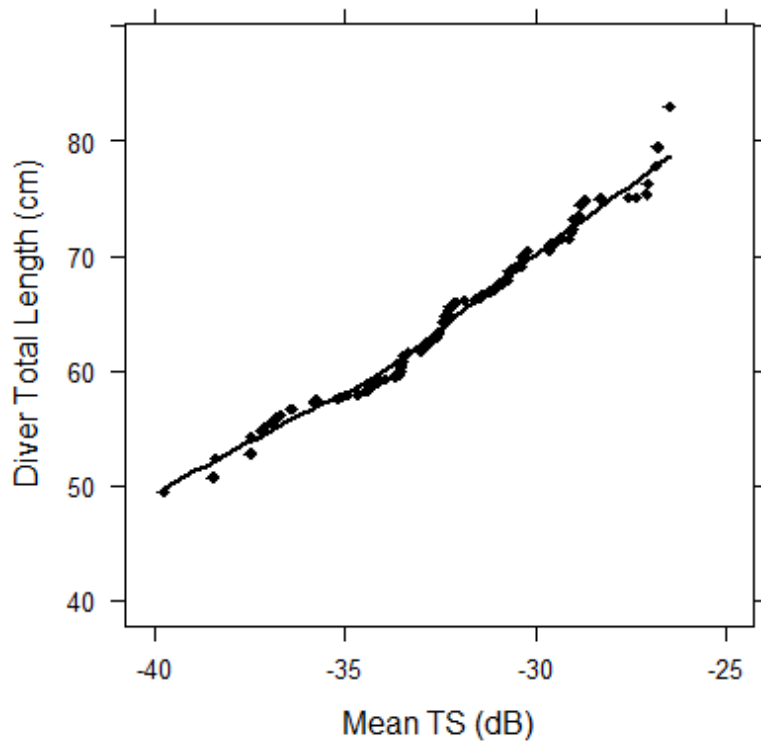
Figure 3.9 Mean TS measurements from tracked fish scaled by the length data from divers resulting in a TS-Length equation for the LCW FSA of TS= 27.6log10(L)-147.32 (R^2 =0.98)(n=193).

3.7 References

Archer, S.K., Heppell, S.A., Semmens, B.X., Pattengill-Semmens, C.V., Bush, P.G., Mccoy, C.M. and Johnson, B.C., 2012. Patterns of color phase indicate spawn timing at a Nassau grouper *Epinephelus striatus* spawning aggregation. *Current Zoology*, *58*(1), pp.73-83.

Balk, H.and Lindem, T. 2006. Sonar 4, Sonar 5, Sonar 6 e Post-processing Systems. Operator Manual. Lindem Data Acquisition, Oslo.

Biosonics. 2004. DTX calibration 2e.doc.rev.1
http://www.biosonicsinc.com/doc_library/docs/DTXcalibration2e.pdf

Biosonics. 2010. User Guide; Visual Acquisition 6, version 6.0

Bush P.G., Lane, E.D., Ebanks-Petrie, G.C,, Luke, K.I., Johnson, B.R., McCoy, C.R., Bothwell, J.O., Parsons, E.U. 2006. The Nassau grouper spawning aggregation fishery of the Cayman Islands – an historical and management perspective Proc. - *Gulf Caribbean Fish Institute.* Univ. Miami 57:515–524.

Cayman Islands Government. 2016. The National Conservation (General) Regulations Part 2.6 http://www.gov.ky/portal/pls/portal/docs/1/12326595.PDF

Carpenter, K.E. et al. 2015. *Epinephelus striatus*. The IUCN Red List of Threatened Species 2015: e.T7862A70324790. Downloaded 09 December 2015

Colin, P.L., 1992. Reproduction of the Nassau grouper, Epinephelus striatus (Pisces: Serranidae) and its relationship to environmental conditions. *Environmental Biology of Fishes*, *34*(4), pp.357-377.

Demer, D., Berger, L., Bernasconi, M., Bethke, E., Boswell, K., Chu, D., Domokos, R.: Calibration of acoustic instruments. *ICES Cooperative Research Report* (326), 133 (2015)

Domeier, M.L. and Colin, P.L., 1997. Tropical reef fish spawning aggregations: defined and reviewed. *Bulletin of Marine Science*, *60*(3), pp.698-726.

Doonan, I.J., Bull, B. and Coombs, R.F., 2003. Star acoustic surveys of localized fish aggregations. *ICES Journal of Marine Science*, *60*(1), pp.132-146.

Doray, M., Josse, E., Gervain, P., Reynal, L. and Chantrel, J., 2007. Joint use of echosounding, fishing and video techniques to assess the structure of fish aggregations around moored Fish Aggregating Devices in Martinique (Lesser Antilles). *Aquatic Living Resources*, *20*(4), pp.357-366.

Ehrhardt, N.M. and Deleveaux, V.K., 2007. The Bahamas' Nassau grouper (Epinephelus striatus) fishery–two assessment methods applied to a data–deficient coastal population. *Fisheries Research*, *87*(1), pp.17-27.

Foote, K. G., Knudsen, H. P., Vestnes, G., MacLennan, D. N., and Simmonds. E. J. 1987. Calibration of acoustic instruments for fish density estimation: a practical guide. Cooperative Research Report. *Const. Int. Explor. Mer*, *144*, 69 pp

Froese, R, Pauly, D. 2016. FishBase. World Wide Web electronic publication. www.fishbase.org, version (10/2016).

Gascoigne, J. 2002. Nassau grouper and queen conch in the Bahamas: Status and management options. Report by MacAlistair Elliot and Partners, Ltd. (MEP) to the Bahamas Reef Environment Educational Foundation (B.R.E.E.F.). http://www.macalisterelliott.com/media/reports/1651R02C.pdf

Gauthier, S. and Horne, J.K., 2004. Potential acoustic discrimination within boreal fish assemblages. *ICES Journal of Marine Science*, *61*(5), pp.836-845.

Gibson, J. 2007. Managing a Nassau Grouper Fishery–A Case Study from Belize. In *Proceedings of the Gulf and Caribbean Fisheries Institute* (Vol. 60, pp. 1-2).

Gledhill, C.T., Lyczkowski-Shultz, J., Rademacher, K., Kargard, E., Crist, G. and Grace, M.A., 1996. Evaluation of video and acoustic index methods for assessing reef-fish populations. *ICES Journal of Marine Science*, *53*(2), pp.483-485.

Guillard, J., Lebourges-Dhaussy, A. and Brehmer, P., 2004. Simultaneous Sv and TS measurements on Young-of-the-Year (YOY) freshwater fish using three frequencies. *ICES Journal of Marine Science*, *61*(2), pp.267-273.

Heppell, S.A., Semmens, B.X., Archer, S.K., Pattengill-Semmens, C.V., Bush, P.G., McCoy, C.M., Heppell, S.S. and Johnson, B.C., 2012. Documenting recovery of a spawning aggregation through size frequency analysis from underwater laser calipers measurements. *Biological Conservation*, *155*, pp.119-127.

ICES Cooperative Research Report No. 238. 2000. Report on Echo Trace Classification. David G. Reid (ed). Copenhagen, Denmark.

Johannes, R.E., 1978. Reproductive strategies of coastal marine fishes in the tropics. *Environmental biology of fishes*, *3*(1), pp.65-84.

Johannes, R.E., Squire, L., Graham, T., Sadovy, Y. and Renguul, H., 1999. Spawning aggregations of groupers (Serranidae) in Palau. *The Nature Conservancy Marine Research Series Publication*, *1*, pp.1-144.

Johnston, S.V., Rivera, J.A., Rosario, A., Timko, M.A., Nealson, P.A. and Kumagai, K.K., 2006. Hydroacoustic evaluation of spawning red hind (Epinephelus guttatus) aggregations along the coast of Puerto Rico in 2002 and 2003.

Jones, D.T., Wilson, C.D., De Robertis, A., Rooper, C.N., Weber, T.C. and Butler, J.L., 2012. Evaluation of rockfish abundance in untrawlable habitat: combining acoustic and complementary sampling tools. *Fishery Bulletin*, *110*(3), pp.332-343.

Kadison, E., Nemeth, R.S., Blondeau, J., Smith, T. and Calnan, J., 2010. Nassau grouper (Epinephelus striates) in St. Thomas, US Virgin Islands, with evidence for a spawning aggregation site recovery. In *Proceedings of the Gulf and Caribbean Fisheries Institute* (Vol. 62, pp. 273-279). Gulf and Caribbean Fisheries Institute, c/o Harbor Branch Oceanographic Institution, Inc. Fort Pierce FL 34946 United States.

Kellner, J.B., Tetreault, I., Gaines, S.D. and Nisbet, R.M., 2007. Fishing the line near marine reserves in single and multispecies fisheries. *Ecological Applications*, *17*(4), pp.1039-1054.

Knudsen, F.R., Hawkins, A., McAllen, R. and Sand, O., 2009. Diel interactions between sprat and mackerel in a marine lough and their effects upon acoustic measurements of fish abundance. *Fisheries Research*, *100*(2), pp.140-147.

Kobara, S. and Heyman, W.D., 2008. Geomorphometric patterns of Nassau grouper (Epinephelus striatus) spawning aggregation sites in the Cayman Islands. *Marine Geodesy*, *31*(4), pp.231-245.

Kracker, L., 2007. Hydroacoustic surveys: A non-destructive approach to monitoring fish distributions at National Marine Sanctuaries. NOAA Technical Memorandum NOS NCCOS 66

Love, R.H., 1971. Measurements of fish target strength: a review. *Fish. Bull,* *69*(4), pp.703-715.

Løland, A., Aldrin, M., Ona, E., Hjellvik, V. and Holst, J.C., 2007. Estimating and decomposing total uncertainty for survey-based abundance estimates of Norwegian spring-spawning herring. *ICES Journal of Marine Science*, *64*(7), pp.1302-1312.

Lilja, J., Marjomäki, T.J., Jurvelius, J., Rossi, T. and Heikkola, E., 2004. Simulation and experimental measurement of side-aspect target strength of Atlantic salmon (Salmo salar) at high frequency. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(11), pp.2227-2236.

Machias, A. and Tsimenides, N., 1995. Biological factors affecting the swimbladder volume of sardine (Sardina pilchardus). *Marine biology*, *123*(4), pp.859-867.

Murphy, H.M. and Jenkins, G.P., 2010. Observational methods used in marine spatial monitoring of fishes and associated habitats: a review. *Marine and Freshwater Research*, *61*(2), pp.236-252.

Nemeth, R.S., 2005. Population characteristics of a recovering US Virgin Islands red hind spawning aggregation following protection. *Marine Ecology Progress Series*, *286*, p.81.

Nielsen, J.R. and Lundgren, B., 1999. Hydroacoustic ex situ target strength measurements on juvenile cod (Gadus morhua L.). *ICES Journal of Marine Science*, *56*(5), pp.627-639.

Ona, E. and Mitson, R.B., 1996. Acoustic sampling and signal processing near the seabed: the deadzone revisited. *ICES Journal of Marine Science*, *53*(4), pp.677-690.

Parker-Stetter, S.L. Rudstam, L.G., Sullivan, P.J., Warner, D.M.,2009. Standard operating procedures for fisheries acoustic surveys in the Great Lakes. Great Lakes Fisheries Commission Special Publication 09-01

Rivera, J, Kellison, T.O., Appeldoorn, R.S., Schärer, M.I., Nemeth, M.I., Rowell, T., Mateos, D.A., Nealson, P.A. 2011. Detection of Mona Island and Abrir La Sierra, Puerto Rico Red

Hind (Epinephelus guttatus) 1 m Off the Bottom with Hydroacoustic Techniques.. In: ***Proc. - Gulf Caribb. Fish. Inst.*** Univ. Miami 63:143-148.

Rose, G.A., 2009. Variations in the target strength of Atlantic cod during vertical migration. ***ICES Journal of Marine Science***, *66*(6), pp.1205-1211.

Rudstam, L.G., Parker, S.L., Einhouse, D.W., Witzel, L.D., Warner, D.M., Stritzel, J.L., Parrish, D.L. and Sullivan, P.J., 2003. Application of in situ target-strength estimations in lakes: examples from rainbow-smelt surveys in Lakes Erie and Champlain. ***ICES Journal of Marine Science***, *60*(3), pp.500-507.

Ryan, T.E., Kloser, R.J. and Macaulay, G.J., 2009. Measurement and visual verification of fish target strength using an acoustic-optical system attached to a trawlnet. ***ICES Journal of Marine Science***, *66*(6), pp.1238-1244.

Sadovy, Y., Domeier, M. 2005. Are aggregation fisheries sustainable? Reef fish fisheries as a case study. ***Coral Reefs*** *24*:254-262

Sadovy de Mitcheson, Y.,Cornish, A., Domeier, M., Colin, P. L., Russell, M., and Lindeman, K. C. 2008. A global baseline for spawning aggregations of reef fishes. ***Conservation Biology***, *22*(5), 1233-1244.

Sadovy, Y, Colin, P.L. (eds). 2012. Reef fish spawning aggregations: biology, research and management. Springer, Berlin, pp 331-369

Sadovy de Mitcheson, Y. and Erisman, B. 2012. Chapter 8. Fishery and biological implications of fishing spawning aggregations, and the social and economic importance of aggregating fishes, p. 225-284 in: Sadovy Y, Colin PL (eds) (2012) Reef fish spawning aggregations: biology, research and management. Springer, Berlin, 644pp

Sadovy de Mitcheson, Y. 2016. Mainstreaming Fish Spawning Aggregations into Fishery Management Calls for a Precautionary Approach. ***BioScience*** *66*.4: 295-306.

Sala, E., Ballesteros, E. and Starr, R.M., 2001. Rapid decline of Nassau grouper spawning aggregations in Belize: fishery management and conservation needs. ***Fisheries***, *26*(10), pp.23-30.

Sawada, K., Furusawa. M., Williamson. N.J., 1993. Conditions for the precise measurement of fish target strength in situ. ***Journal of the Marine Acoustical Society***. *20*; 15–21.

Semmens, B.X., Bush, P.H., Heppell, S.C., Johnson, B.R, McCoy, C.R., Pattengill-Semmens, C.H., Whaylen, L.E. 2007. Charting a course for Nassau grouper recovery in the Caribbean: what we've learned and what we still need to know. *In **Proc Gulf Caribb Fish Inst*** 2008 (Vol. 60, pp. 607-609).

Simmonds, J. and MacLennan, D.N., 2005. *Fisheries acoustics: theory and practice*. John Wiley and Sons.

Smith, C.L., 1972. A spawning aggregation of Nassau grouper, *Epinephelus striatus* (Bloch). ***Transactions of the American Fisheries Society***, *101*(2), pp.257-261.

Soule, M., Barange, M. and Hampton, I., 1995. Evidence of bias in estimates of target strength obtained with a split-beam echo-sounder. *ICES Journal of Marine Science*, *52*(1), pp.139-144.

Stallings, C.D. 2008. Indirect effects of an exploited predator on recruitment of coral reef fishes. *Ecology, 89*: 2090–2095. doi:10.1890/07-1671.1

Stallings, C.D., 2009. Predator identity and recruitment of coral-reef fishes: indirect effects of fishing. *Marine Ecology Progress Series*, *383*, pp.251-259.

Starr, R.M., Sala, E., Ballesteros, E. and Zabala, M., 2007. Spatial dynamics of the Nassau grouper *Epinephelus striatus* in a Caribbean atoll. *Marine Ecology Progress Series*, *343*, pp.239-249.

Starr, R.M., Wendt, D.E., Barnes, C.L., Marks, C.I., Malone, D., Waltz, G., Schmidt, K.T., Chiu, J., Launer, A.L., Hall, N.C. and Yochum, N., 2015. Variation in responses of fishes across multiple reserves within a network of marine protected areas in temperate waters. *PloS One*, *10*(3), p.e0118502.

Taylor, J.C., Eggleston, D.B. and Rand, P.S. 2006. Nassau grouper (*Epinephelus striatus*) spawning aggregations: hydroacoustic surveys and geostatistical analysis. In: Emerging technologies for reef fisheries research and management. Seattle, WA, NOAA, pp. 18-25. (NOAA Professional Paper NMFS, 5)

Thomas, G.L. and Thorne, R.E., 2003. Acoustical-optical assessment of Pacific herring and their predator assemblage in Prince William Sound, Alaska. *Aquatic Living Resources*, *16*(3), pp.247-253.

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C., 2011. Underwater acoustics for ecosystem-based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, *442*, pp.285-301.

Utne, K.R. and Ona, E., 2006. Acoustic extinction in dense herring layers, measured from a bottom-mounted transducer. ICES Document CM/2006/I: 10. 2006:17

Whaylen, L., Pattengill-Semmens, C.V., Semmens, B.X., Bush, P.G. and Boardman, M.R., 2004. Observations of a Nassau grouper, Epinephelus striatus, spawning aggregation site in Little Cayman, Cayman Islands, including multi-species spawning information. *Environmental Biology of Fishes*, *70*(3), pp.305-313.

Whaylen L, Bush, P.H., Johnson, B.R., Luke, K.E., McCoy, C.R., Heppell, S.C., Semmens, B.R., Boardman, M.A. 2006. Aggregation dynamics and lessons learned from five years of monitoring at a Nassau grouper (Epinephelus striatus) spawning aggregation in Little Cayman, Cayman Islands, BWI. *Proc. - Gulf Caribb. Fish. Inst*. Univ. Miami 57:1-14

Winfield, I.J., Emmrich, M., Guillard, J., Mehner, T. and Rustadbakken, A., 2011. WISER Deliverable 3.4-3: Guidelines for standardisation of hydroacoustic methods. *WISER Project Newsletter*, *2*, pp.4-4.

Zhao, X. and Ona, E., 2003. Estimation and compensation models for the shadowing effect in dense fish aggregations. *ICES Journal of Marine Science*, *60*(1), pp.155-163.

4 Chapter 4. Hydroacoustics as a tool to examine the effects of protection and habitat type on marine fish communities

4.1 Abstract

Although hydroacoustic technologies are widely used in fisheries research, there are few studies examining the effects of Marine Protected Areas (MPAs) using hydroacoustic methods. We help close this knowledge gap by using hydroacoustics to examine the effects of habitat type and closure to fishing, on fish populations in the Cabo Pulmo National Park (CPNP) MPA. Fish density (fish/ha) was significantly higher for transects within the CPNP compared to those outside in non-MPA control areas (447 ± 141 S.E.M vs 112 ± 19, respectively), and higher still over reef-specific transects inside the park (5388 ± 1282). Largest mean fish size was also found over reef-specific transects, followed by mixed habitats inside the park, then the rocky and finally sandy control sites outside the park. Acoustic fish biomass estimates differed significantly between sites ranging from 1.88 t/ha over the CPNP reef-specific transects to 0.01 t/ha at the sandy control site. Acoustic estimates of fish biomass did not differ significantly from those calculated using Underwater Visual Census (UVC) data, although densities did, due to higher fish numbers in the smallest UVC size classes. This study is a valuable starting point in demonstrating the utility of hydroacoustics for the assessment of fish in coastal MPAs.

4.2 Introduction

Coastal marine environments face ever-increasing threats from growing anthropogenic pressures (Crain et al. 2009). Marine Protected Areas (MPAs) have been suggested as one of a suite of spatial management tools attempting to reduce the pressures posed by these threats on marine life and habitats (Beverton and Holt, 1957). Whilst MPAs may be designated for a variety of reasons (Boersma and Parrish, 1999; Allison et al. 1998) one common objective is the protection of exploited fish populations (Sale et al. 2005). Consequently, there have been many studies on the efficacy of MPAs in protecting fish and recent meta-analyses report positive increases of fish density, diversity, body-size and biomass (Halpern, 2003; Claudet et al. 2008; García-Charton et al. 2008; Lester et al. 2009; Sciberras et al. 2015; Starr et al. 2015) within MPAs. Developing suitable fish population monitoring programmes for MPAs is, however, often a difficult task (Hill et al. 2014).

Most studies on the response of fish populations to different levels of fishing intensity or management regime overlook pelagic species and tend to focus on less mobile demersal species for which there are stronger links with bottom habitat types (Claudet et al. 2008; Stewart et al. 2009; Johnson et al. 2013). Such studies tend to use survey techniques such as trapping, fishing, camera recordings and Underwater Visual Census (UVC). Fish survey methods that provide more detail on the mid-water component within MPAs may also reveal how pelagic species and the benthic-pelagic coupling respond to protection, an area of research that has limited data (Game et al. 2009; Stewart et al. 2009; Grober-Dunsmore et al. 2008). In this respect, active hydroacoustics has the advantage that it can sample almost the entire water column (Yurista et al. 2014), whereas UVC is focussed predominantly on demersal species (i.e. from the seabed to a given height above it). Hydroacoustics can also cover a much greater area per unit of time, allowing large spatial scales to be studied which may be necessary to sample highly mobile species. The relatively fast data acquisition of hydroacoustic methods also adds to the time-saving (and therefore often cost-saving) benefits when compared to alternative fish survey methods (Jones et al. 2012), and data are digitally recorded immediately following acquisition (Trenkel et al. 2011). Hydroacoustic fish survey methods also have the advantage that they are non-destructive in nature and are not hampered by issues such as water clarity, strong currents or diver depth limits. Hydroacoustic methods do, however, require groundtruthing to gain species-specific information and for the most accurate calculations of fish lengths and weight (Mackinson et al. 2004).

Although hydroacoustic surveys offer many advantages over other fish survey methods, we are unaware of any published research using hydroacoustics to evaluate the effects of both protection regime and habitat type on marine fish populations. Chapter 3 (Egerton et al. 2017) used hydroacoustics to locate and quantify Nassau grouper spawning aggregations within MPAs, but did not examine the effects of protection *per se*. Polunin et al. (2009) found that acoustic surveys can provide a cost-efficient method of assessing fish biomass within an MPA in comparison to other methods (baited traps, baited video and trammel nets), but did not aim to use these different methods to compare fish communities inside vs outside their protected study area. Similarly, Rudershausen et al. (Rudershausen et al. 2010) used acoustics and fish traps in an MPA off the South eastern US coast, with the aim of comparing the two methods rather than examining the effect of protection regime or habitat type on the local fish populations. Habitat type is well understood to have a significant influence on fish community composition and distribution (Ortiz and Tissot, 2012; Miller and Russ, 2014). In order to comprehensively evaluate the effects of marine protection on fish populations, seabed habitat type therefore needs to be taken into account (Lester et al. 2009; Claudet et al. 2006). Most studies demonstrate increases in fish abundance (Luckhurst and Luckhurst, 1978) and biomass (Grigg, 1994; Graham and Nash, 2013) with increasing habitat complexity. Many studies evaluating the effects of MPAs, however, often fail to consider such habitat effects, which, in some cases may mask the effects of protection considerably (Stewart et al. 2009; Miller and Russ, 2014). Both the abiotic and biotic habitat can also have effects on fish assemblages (Thiriet et al. 2016), however here our focus is the abiotic habitat and throughout this study use the word 'habitat' synonymously with 'substrate' (Diaz et al. 2004).

The Cabo Pulmo National Park (CPNP), Baja California Sur, Mexico was established in 1995, with considerable involvement from the local community (Havard et al. 2015), and covers an area of 7,111 hectares (Verutes et al. 2014) (Figure 4.1). A major factor governing the success of an MPA is how well it has been enforced (Edgar et al. 2014), and although only 35% of the CPNP is designated as a 'no take' area, the local community follow and enforce a policy of no-fishing throughout the entire reserve (Jones, 2011). Fourteen years following the creation of the park, UVC surveys reported a 463% increase in fish biomass (Aburto-Oropeza et al. 2011). This is in keeping with a mean biomass increase of 446% reported in a meta-analysis of 55 MPAs globally (Lester et al. 2009). The CPNP is composed of a mixture of habitats with basaltic rocky reef dikes forming long, parallel ridges that run adjacent to shore in the northern

section of the park, while disappearing under the shoreline in the south-central section (Riegl et al. 2007). Isolated coral heads grow on top of these ridges, and the highest amount of coral cover is around 15-20% over central sections (Reyes-Bonilla and Calderon-Aguilera, 1999). Between the rocky reefs the seafloor habitats consist primarily of sand interspersed with sparse boulder fields.

Past fish population surveys of the CPNP (e.g. Aburto-Oropeza et al. 2011) have employed teams of SCUBA divers surveying linear transects along the rocky reefs of the park counting fish and invertebrates, estimating mean sizes of individual and schooling fish. In this study we use a split beam echosounder to conduct hydroacoustic surveys to evaluate the effects of protection from fishing and habitat type by examining the total fish density, total fish biomass and mean fish size within the CPNP in comparison to sites outside the park (Thesis Hypotheses 4 and 5a). Further, the hydroacoustic 'reef-specific' transects that specifically targeted the reefs within the park are compared with the belt transect UVC estimations carried out over corresponding reef sites in the same year (Thesis Hypothesis 3). Finally, we comment on the efficacy of using hydroacoustic surveys to measure the effects of protection and habitat type on fish populations.

4.3 Methods

4.3.1 Field surveys

Hydroacoustic field surveys in and around the CPNP were undertaken during March 2015 during daylight hours in collaboration with a local SCUBA diving company (Cabo Pulmo Divers). All survey protocols were approved by Comisión Nacional de Áreas Naturales Protegidas (CONANP).

To undertake hydroacoustic fish surveys, there needs to be adequate coverage over the survey areas to gain a reliable picture of the local fish distributions. Degree of coverage (Λ) is defined as:

$\Lambda = D/\sqrt{A}$ (Equation 4.1.)

where: D is the cruise track length; and, A is the size of the survey area (Aglen 1989), and for adequate coverage the ratio generally needs to be ≥ 6. This was achieved in all the different survey sites. Control sites outside the park and therefore open to fishing were: 1) Punta Arena (PA), a mainly sandy bottom site located 5 km to the north of the CPNP; and 2) Bajo del Salado (BS), a boulder-reef complex 5 km to the south. High resolution reef-specific surveys were

also undertaken inside the CPNP by running hydroacoustic transects along each discrete reef area (Figure 4.1). These reef areas were located using the local knowledge of the skipper and previous SCUBA monitoring of the sites. On all reef-specific transects, bottom type was confirmed using a towed camera system to ensure that the reefs were being correctly targeted. The towed camera system was also used to identify groups of fish species within transects where possible. Overall the whole hydroacoustic survey campaign took one researcher and one boat operator a total of 8 days to survey the whole 7,111 ha park (with double the necessary coverage), the two control sites and the final reef-specific survey within the CPNP.
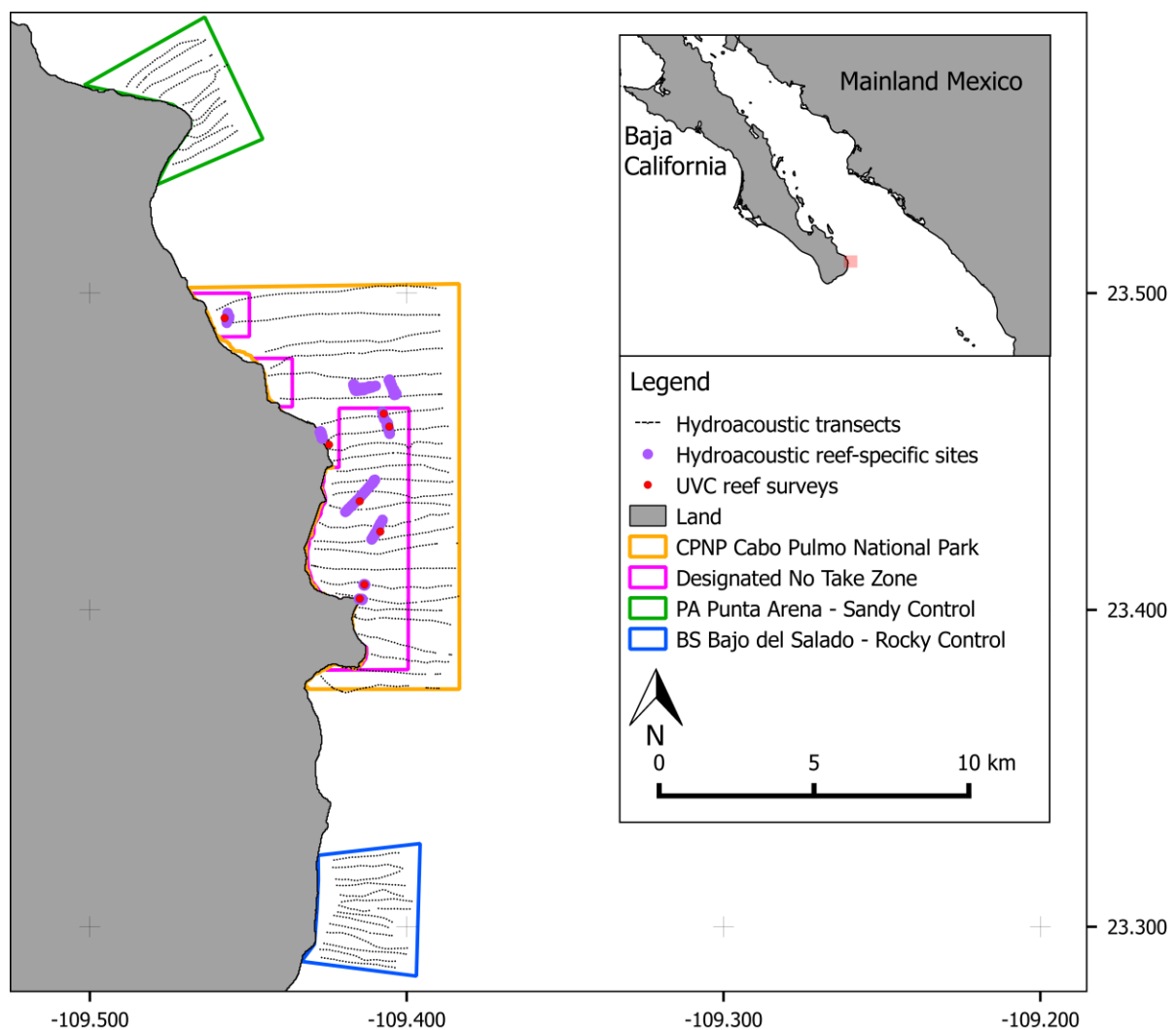
Figure 4.1 Location of the survey sites and transect lines at Cabo Pulmo, Baja California Sur, Mexico. Coordinates are in WGS84.

Table 4.1 Descriptions of the survey sites summarising substrate types, mean depths and protection afforded.

Site name	Abbreviation	Habitat/Substrate type	Mean Depth of area Investigated (m) (± S.D)	Protection
Cabo Pulmo Nat. Park	CPNP	Sand, boulders and rocky reefs	72.6 ±23.5	✓
Punta Arena	PA	Mainly sand	86.9 ±14.9	✗
Bajo Del Salado	BS	Boulder-reef complex	59.8 ±24.9	✗
Reef-specific	REEFS	Rocky basaltic reefs	12.3 ±4.4	✓

4.3.2 Acoustic Equipment

A Biosonics® DTX Split beam echosounder with a 200 kHz transducer was used for the surveys, pole mounted over the side of the survey vessel with the transducer face 1 m under the water surface. Acoustic data were georeferenced with an integrated Garmin 17Xhvs GPS, and a laptop computer loaded with Biosonics acquisition software (Visual Acquisition 6). The circular transducer used with this system has a beam-opening angle of 6.8º (3 dB beam width). Pulse duration was 0.4 ms with a specified max ping rate of 10 per second. Survey speed throughout the surveys was kept under 6 knots. Calibration used a standard Biosonics 36mm Tungsten Carbide 200 kHz calibration sphere before the surveys, following the standard methods of Foote et al. (1987).

4.3.3 Acoustic Data processing

Data were collected as DT4 files and then converted and post-processed with the Sonar5 Pro (ver 6.0.4)- software package (Balk and Lindem 2006). Analysis followed the Software Guided Analysis (SGA) routine based upon the standard operating procedure of Parker-Stetter et al. (2009). Density estimates were calculated by echo integration (EI), which divides the average

reflection from all fish over a specified volume (the volume backscattering coefficient, Sv) by the average backscattering cross section (σ_{bs}) which is derived from the mean echo intensity (Target Strength (TS)) from individual fish (Draštík et al. 2009; Winfield et al. 2011). These TS values were obtained *in situ*. Analyses were based on Single Echo Detections (SED) rather than tracked fish as on some transects there were too few confirmed fish tracks present (likely due to vessel movement and the relatively long pulse duration used) for accurate calculations to be made with EI on some transects. SEDs had a minimum echo length of 0.8 dB, a maximum of 1.2 dB and a maximum angle standard deviation of 0.8 degrees. Multi-peak suppression was set to 'medium' in the Sonar5 (ver 6.0.4) software which requires a dip of 1.5 dB between peaks if the echo is to be rejected. Thresholds of -60 dB for TS values, and -66 dB for Sv values were applied to the data to initially discern fish from other particulate targets such as plankton (Reid, 2000). To ensure that no echoes from the seabed were classified as fish (Ona and Mitson, 1996), a bottom layer of 1 m was applied and any returns from this layer were removed unless they could be clearly identified as fish with definite separation visible between the return and the seabed. Similarly, a surface layer of 1 m was applied to remove surface noise from wind and wave action, on occasion this noise had to be removed to 5 m due to abnormally poor surface conditions which was undertaken manually. The data were processed up to a depth limit of 100 m as beyond this the acoustic signal to noise ratio (SNR) became unacceptably low (Yule et al. 2013). To compensate for changes in echo intensity due to increasing range (R), a Time Varied Gain (TVG) of 40log(R) for TS values and 20log(R) for Sv values was used as recommended in Sonar5 (ver 6.0.4) (Balk and Lindem, 2006). Whole transects were taken as Elementary Sampling Distance Units (EDSU's) to maximize the number of EDSUs with sufficient SED as the source of *in situ* TS for the calculations (Balk and Lindem 2006). To minimize potential spatial autocorrelation, we calculated Sv for the entire esonified water volume (Emmrich et al. 2010). Mean TSs were checked for bias following Sawada et al. (1993) and each transect had a fish per esonified volume (Nv) less than 0.1 (Rudstam et al. 2009).

4.3.4 Acoustic Fish Size

TS is an indicator of fish size but is also influenced by species due to differences in ratios of body size to bladder size (Simmonds and MacLennan, 2005) and swimming behaviour (tilt angle) of the species or individual (Nielsen and Lundgren, 1999). To translate TS into more intuitive length measurements (cm) than the decibel (Boswell et al. 2007) it is converted using empirical TS-length relationships, which often exist for specific groups or species (Kracker

2007). There was a wide diversity of fish species in the area making the use of species-specific TS–length formulas problematic (Coll et al. 2007). Further, for most of the species identified using a towed underwater camera, empirical TS-length formulae are yet to be established. It was therefore necessary to apply the multi-species equation from Love (1971) to convert TS to length. It provides the following conversion:

TS = (19.1 log$_{10}$L) – (0.9 log$_{10}$ f) – 62.0 (Equation. 4.2)

Where TS = target strength detected (dB), L = length (Total Length) of the target (cm), and f = the frequency used. With a transducer frequency of 200 kHz (as used in these surveys) this equation then becomes:

TS = 19.1 log$_{10}$(L) - 64.07 (Equation. 4.3)

Reformulation of Equation 4.3 to gain unknown lengths from known TS therefore becomes:

L= 10^((TS+64.07)/19.1 (Equation. 4.4)

4.3.5 Acoustic Biomass

Total fish biomass was examined as it provides a better measure of productivity than fish density values (Nagelkerken et al. 2012). Commonly in hydroacoustic surveys, the scattering coefficient (sA) presented in terms of an area (m^2/ha) is used as a proxy for biomass (Simmonds and MacLennan, 2005; Boswell et al. 2010). It quantifies the amount of a unit area occupied by fish, considering the water depth (Coll et al. 2007). We compared these sA units between sites and sA values was also compared to linearized sv values (units of m^2/m^3) to ensure differences in depth between sites were not having significant influence on this parameter. To compare hydroacoustic biomass values with those estimated from UVC surveys, it was also necessary to calculate biomass values with units of t/ha following Yurista et al. (2014). To do this, 5 different steps were undertaken: 1) TS distributions (based on SED from -60 dB to -20 dB in 1 dB bins) for each transect were converted to fish length by using the aforementioned multispecies equation from Love (1971). 2) The midpoint of these length bins was then converted to weight by using a generalist W-L equation gained from a meta-analysis of 451 species by Froese (2006): W = a x L^b (constants a = 0.0137 and b = 3.03 and L is Total Length) (Equation. 3). 3) The proportions of the different weights present in each sample were then multiplied by the total density values (# fish/ha) of each transect, giving the number of fish per 5 cm weight class. 4) These values were summed per weight class to give a biomass value in

t/ha. 5) The average biomass value per site was then calculated as the arithmetic mean of all transects within each site.

4.3.6 UVC Surveys

We took advantage of the monitoring program that has been undertaken over the reefs within the CPNP during the months of August and September since 2000 (Aburto-Oropeza et al. 2011; Erisman et al. 2011; Mascareñas-Osorio et al. 2011; Aburto-Oropeza et al. 2015). Under this program, UVC surveys using SCUBA are conducted using the standard methods for visual belt transects (Harmelin-Vivien et al. 1985). A total of six divers count and identify all fishes observed to species level at each reef site. UVC data from 8 reefs (those in the same locations as the hydroacoustic reef surveys) were used for comparison with the hydroacoustic data we collected during the field campaign. At each site, a two-person dive team survey 50 m transects counting and estimating the sizes of all fish and invertebrates, within a 5m wide belt along each transect during two passes. This results in between 4 and 8 replicates per each 250 m^2 total area. Mobile species versus territorial species are surveyed in separate passes, to ensure that the same individuals are only counted once. A table of fish species recorded from the UVC surveys over the reefs, with densities, trophic group, mean sizes and biomasses is provided in Appendix 1.

4.3.7 Statistical analyses

A large school of jacks (*Caranx Sexfasciatus*) (9m high by 25m long) was encountered during the reef-specific acoustic transects, this stochastic event created a significant outlier in the data (increasing mean acoustic 'reef-specific' density values by 20%). There are also many difficulties in calculating density estimates for dense fish schools due to sound attenuation (Simmonds and MacLennan 2005). We therefore excluded this from all further analyses meaning our estimates of fish density and biomass for the reef-specific surveys are conservative. Mean values (± S.E.M) of fish density, biomass and size were calculated from the two surveys inside the park as there was no significant difference between these surveys for any of the parameters. This also highlights the repeatability of the method we employed. Biomass values derived from TS values were compared with sA values via ordinary least squares (OLS) regression. To compare fish density, biomass, sA and length data between the different sites, Welch's one-way ANOVAs were used. Following each ANOVA, Tukey's *post-hoc* multiple comparisons were performed to determine where any significant differences

between sites occurred. If assumptions of normality or equal variance were not met, then the data were first $\log_{10}$ transformed. If data transformation did not address the violations of the assumptions of normality, a Kruskal-Wallis test was used on the non-transformed data followed by Dunn's *post-hoc* tests. Two-sample T-tests (or where necessary the non-parametric equivalent Mann-Whitney) were then also used to test differences in mean fish density, biomass, sA and fish size between the CPNP transects and all control transects. Mean fish size and biomass data from SCUBA UVC surveys undertaken in 2015 were compared with our reef-specific fish data using Mann-Whitney tests and fish size class frequency distributions by a 2 sample Kolmogorov–Smirnov test following reassignment of acoustic data to the fish size classes given by UVC surveys.

4.4 Results

4.4.1 Number of fish in CPNP vs control areas

There were significantly different acoustic mean densities of fish per hectare between all sites (ANOVA, $F_{3,68}$ = 43.9, P < 0.001). Greatest acoustic fish density was present in the 'reef-specific' surveys (5388 ± 1282 S.E.M fish/ha) within the park. These were an order of magnitude higher than numbers gained during the standard acoustic survey transects within the CPNP which combined both reef, rocky and sandy habitats (447 ± 141 fish/ha), and higher still in comparison to the control areas (PA = Punta Arena, the sandy control site: 130 ± 40, and BS = Bajo del Salado, the rocky control site: 99 ± 17 fish/ha). Pairwise comparisons between sites showed that CPNP acoustic transects had significantly higher fish density than BS whilst there was no significant difference between BS and PA, or the CPNP and PA (Table 4.2, Figure 4.2). A comparison of fish density inside (CPNP transects excluding the reef-specific surveys) versus outside the reserve (PA and BS) showed fish density was four times higher within the park (T-test, T_{51} = 3.19, P = 0.002). Density data are also shown spatially in Appendix 2 (Figure 4.7).

Table 4.2 Results of Tukey HSD post-hoc tests examining the differences in fish density ($\log_{10}$ fish number/ha) between sites. PA = Punta Arena (Sandy control), BS = Bajo del Salado (Rocky control), CPNP = Cabo Pulmo National Park, Reefs = Reef-specific hydroacoustics transects within the CPNP.

Site	PA		Reefs		CPNP	
	T	P	T	P	T	P
Reefs	9.18	<0.001	-	-	-	-
CPNP	2.33	0.102	8.53	<0.001	-	-
BS	0.06	1.00	10.20	<0.001	2.71	<0.001
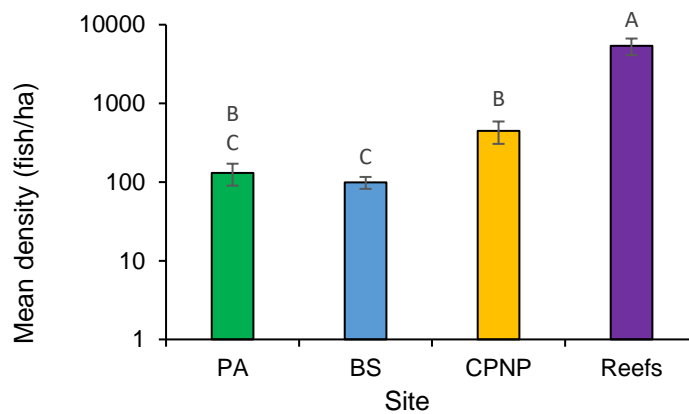
Figure 4.2 Mean numbers of fish/hectare (plotted on a $\log_{10}$ scale) at the different sites surveyed Error bars show ± S.E.M. Bars that share letters are not significantly different from one another. PA = Punta Arena (Sandy control), BS = Bajo del Salado (Rocky control), CPNP = Cabo Pulmo National Park, Reefs = Reefs within the CPNP.

4.4.2 Acoustic Fish Biomass
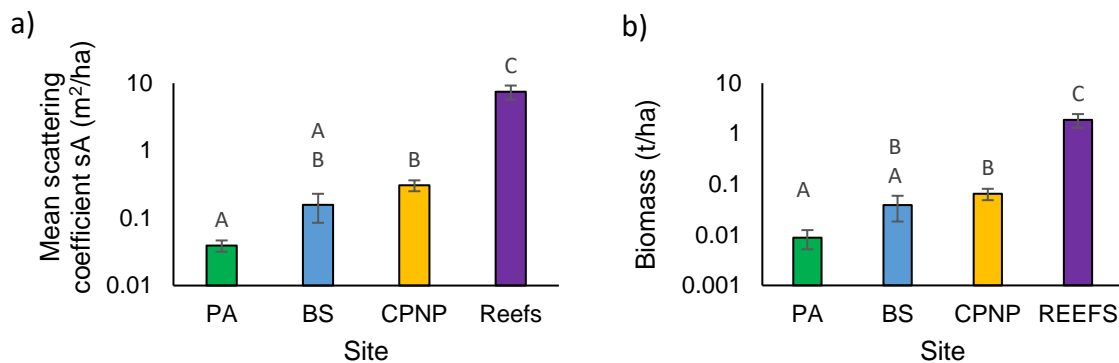
Figure 4.3 Mean values of the area scattering coefficient (sA) (a) and fish biomass (b) (both plotted on a $\log_{10}$ scale) across the different sites. Error bars show $\pm$ S.E.M. Bars that share letters are not significantly different from one another. PA = Punta Arena (Sandy control), BS = Bajo del Salado (Rocky control), CPNP = Cabo Pulmo National Park, Reefs = Reefs within the CPNP.

There was a significant correlation between the mean area scattering coefficient (sA) biomass proxy values (the amount of backscattered energy from fish over a given area) and the calculated acoustic fish biomass values for each transect (tonnes/ha) at all sites (Pearson correlation = 0.936, $P < 0.001$, $R^2 = 0.876$). The sA values were also compared to the volume scattering coefficient, linearized sv values (units of m^2/m^3) (Pearson correlation = 0.967, $P < 0.001$, $R^2 = 0.934$) demonstrating differences in depth (i.e. volumes esonified) between sites were not having significant influence on the calculation of sA. Mean sA values recorded at all sites were significantly different (Kruskal-Wallis, $H_3 = 41.79$, $P < 0.001$) (Table 4.3, Figure 4.3) as were the $\log_{10}$ transformed biomass values (t/ha) between sites (ANOVA, $F_{3,68} = 21.75$, $P < 0.001$) (Figure.3b). Pairwise comparisons between sites showed that reef-specific transects had a significantly higher fish biomass than all other sites and the CPNP and BS had similar biomass values of biomass as did PA and BS (see Table 4.3b below, for details). When both PA and BS were examined together as a general "outside MPA" group, and compared to the CPNP, the biomass values in the CPNP were 273% higher and significantly different for both sA and tonnes per hectare values (Mann-Whitney, $W = 920.5$, $P < 0.001$; $T_{51} = 3.81$, $P < 0.001$, respectively). Fish biomass data (sA) are also shown spatially in Appendix 2 Figure 4.8.

Table 4.3 a). Results of Dunn's post-hoc tests examining the differences in sA between sites and, b) Results of Tukey HSD post-hoc tests examining the differences in biomass ($\log_{10}$ biomass t/ha) between sites. PA = Punta Arena (Sandy control), BS = Bajo del Salado (Rocky

a) Site	PA		Reefs		CPNP	
	Z	P	Z	P	Z	P
Reefs	5.99	<0.001	-	-	-	-
CPNP	3.46	<0.001	3.5	<0.001	-	-
BS	1.6	0.125	4.88	<0.001	1.94	0.05

b) Site	PA		Reefs		CPNP	
	T	P	T	P	T	P
Reefs	7.46	<0.001	-	-	-	-
CPNP	3.8	0.002	4.92	<0.001	-	-
BS	1.84	0.264	6.23	<0.001	2.03	0.19

4.4.3 Acoustic Fish Size

There was a significant difference in the mean size of fish (estimated via acoustics) between all sites (Kruskal-Wallis, $H_3 = 258.22$, $P < 0.001$) (Figure 4.5a). Comparisons between sites revealed that the mean size of fish at PA (mean size 6.02 cm ± 0.62) and those over the reef-specific transects (mean size 14.78 cm ± 0.2) were both significantly different from all other sites (Table 4.4) whilst the mean size of fish inside the CPNP (mean size 11.4 cm ± 0.69) and at BS (mean size 9.5 cm ± 0.71) were not significantly different.

Table 4.4 Results on Dunn's post-hoc tests on mean fish size as estimated via acoustics between sites. PA = Punta Arena (Sandy control), BS = Bajo del Salado (Rocky control), CPNP = Cabo Pulmo National Park, Reefs = Reef-specific hydroacoustics transects within the CPNP.

Site	PA		Reefs		CPNP	
	Z	P	Z	P	Z	P
Reefs	12.27	<0.001	-	-	-	-
CPNP	5.99	<0.001	10.19	<0.001	-	-
BS	4.56	<0.001	4.66	<0.001	0.16	0.88

4.4.4 Comparing hydroacoustics and Underwater Visual Census (UVC) estimates over the CPNP reefs

There was no significant difference between hydroacoustic median biomass values and UVC median biomass values (Mann-Whitney, W = 46, P = 0.392), but fish density values between these two methods of fish survey were significantly different (Mann-Whitney, W = 23, P = 0.016) (Figure 4.4a, b).
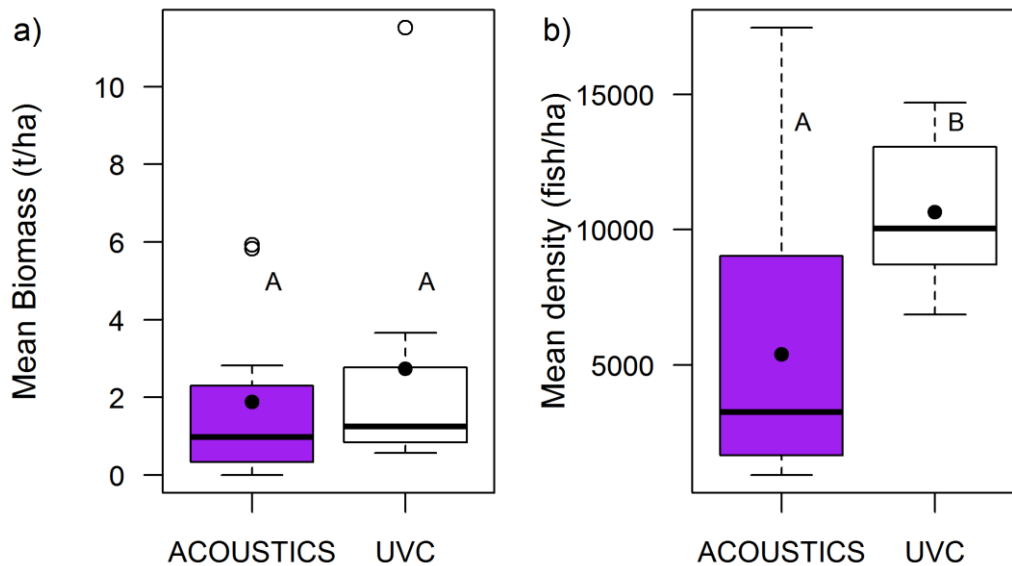
Figure 4.4 a) Fish biomass and b) density estimates over the reefs of the Cabo Pulmo National Park (CPNP) from hydroacoustics (purple boxes) and from Underwater Visual Census (UVC) (white boxes) surveys in the same year (2015). Boxes that share letters within plots are not significantly different from one another. Box plots show mean values (black circle), median values (solid horizontal line), and the lower and upper ends of the box are the 25% and 75% quartiles respectively. The whiskers indicate 1.5 times the inter-quartile range and points beyond this range are shown by empty circles.

Fish size over the reef-specific transects was significantly higher from the hydroacoustic survey estimates (median = 8cm) than the UVC surveys (median = 3cm) (Mann-Whitney, W = $1.16e^9$, P < 0.001). Further, a significant difference in the shape of the size class distributions was also detected between the two methods (Kolmogorov–Smirnov, KS = 0.309, P = 0.023), with greater number of fish in the smaller with categories with UVC surveys (Figure 4.5b).
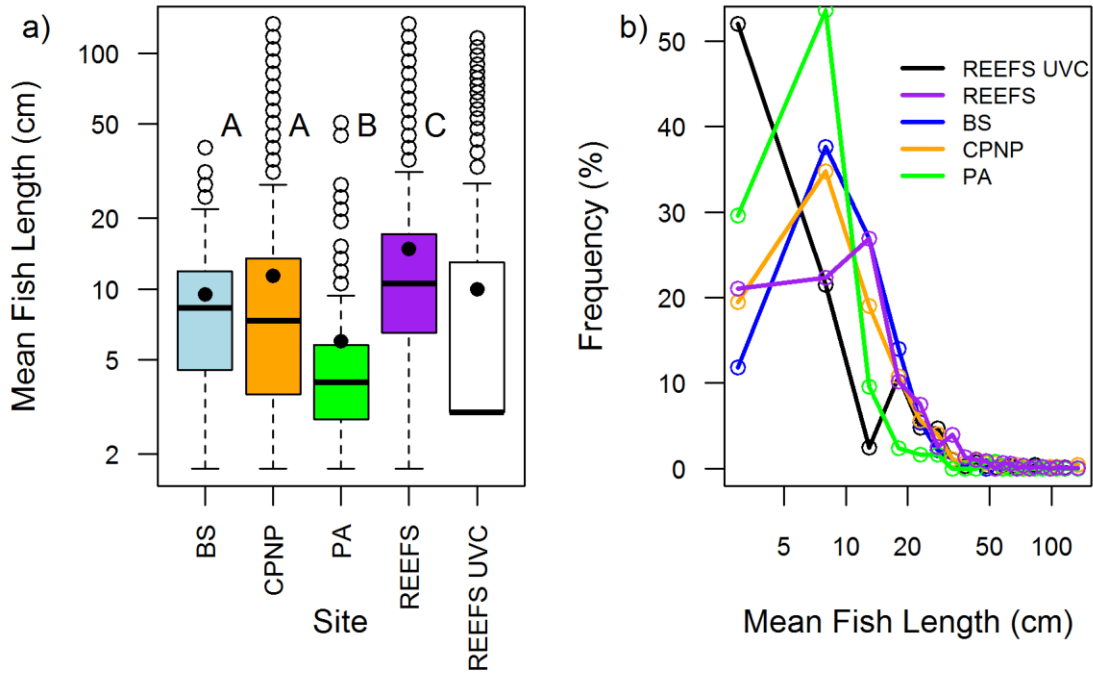
Figure 4.5 a) Mean fish size (plotted on a log10 scale) at the different survey sites. Sizes in cm gained from converting TS to length through application of the Love (1971) formula. Fish size data from Underwater Visual Census (UVC) surveys of the same reefs in 2015 are shown in the white box and no whisker is present due to the median being in the lowest size class. Bars that share letters are not significantly different from one another (UVC data not included in comparisons). Box plots show mean values (black circle), median values (solid horizontal line), and the lower and upper ends of the box are the 25% and 75% quartiles respectively. The whiskers indicate 1.5 times the inter-quartile range and points beyond this range are shown by empty circles. b) Mean proportions of fish sizes at the different survey sites. Colours in b relate to the sites in a. Data on fish length is plotted on a log10 scale. PA = Punta Arena (Sandy control), BS = Bajo del Salado (Rocky control), CPNP = Cabo Pulmo National Park, Reefs = Reef-specific hydroacoustics transects within the CPNP.

Fish density, biomass and mean fish length data from both the hydroacoustic and UVC surveys are summarised in Figure 4.6. This shows how protected sites have a larger mean size, density and biomass, and how although UVC recorded more fish than the acoustic reef surveys, but similar biomass due to a lower mean size of fish.
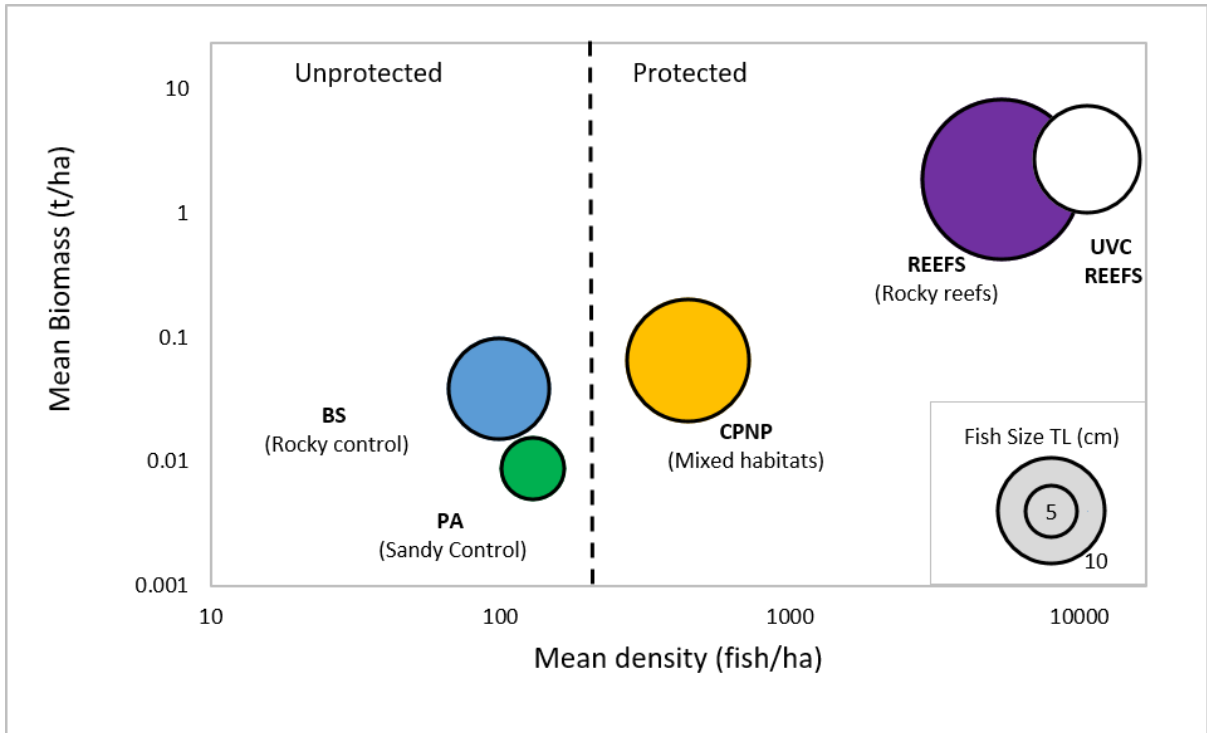
Figure 4.6 Bubble plot summarising the main findings in this study (data are plotted on log10 scale). PA = Punta Arena (Sandy control site), BS = Bajo del Salado (Rocky control site), CPNP = Cabo Pulmo National Park, Reefs = Reefs within the CPNP, UVC Reefs = Data for fish collected over the CPNP reefs using Underwater Visual Census (UVC).

4.5 Discussion

The literature on both the effects of MPAs as management tools and hydroacoustics for fish surveys is plentiful. To our knowledge, however, this is the first study attempting to investigate MPA effects using hydroacoustics, despite the recognised potential in doing so (Kracker, 2007). Our hydroacoustic surveys showed that there were more, larger fish inside the CPNP than in control areas, outside the park (Thesis Hypothesis 4). This is in keeping with most meta-analyses that have reported increased fish density, size and biomass inside MPAs (e.g. Starr et al. 2007). The differences in fish community measures (density, biomass and size classes) between the control sites also emphasizes the importance of habitat type in determining fish community composition inside versus outside the CPNP (Thesis Hypothesis 5a) (Friedlander et al. 2003; Friedlander et al. 2007). It is noteworthy that the greatest fish density, biomass values and mean size were associated with the basaltic rocky reefs within the CPNP. The rocky boulder complex control site (BS) also had more large fish than the sandy bottom control site (PA). Both results highlight the importance of protection in addition to habitat type in determining increased fish density, biomass and size structure, and the utility of using hydroacoustic methods to survey large areas without the associated problems of extractive fishing methods.

4.5.1 Hydroacoustic considerations

Deriving fish density, size and subsequent biomass estimates from applying acoustic *in situ* Target Strength (TS) is not an exact science (Rudstam et al. 2009; Simmonds and MacLennan 2005) as TS is known to vary due to factors such as fish species, aspect, behaviour, condition and maturity (Love, 1977; Frouzova et al. 2005). However, in a mixed species assemblage such as this, it was the only viable option and the same approach has been used previously in similar situations (Wanzenböck et al. 2003; Boswell et al. 2007; Boswell et al. 2010; Zenone et al. 2017). Further imprecision in fish size is likely to have been introduced by using the standard Love (1971) formula in the conversion of TS to fish size. This may not be suitable for all the fishes within the survey area, but provides a consistent relative scale to describe biological sizes of the fish community across all of the sample sites (Boswell et al. 2007; Pollom and Rose, 2015; Wheeland and Rose, 2015). The values derived from this should therefore be considered as an approximation as species-specificity was not possible due to the highly diverse fish community and lack of species-specific conversion formulae (Simmonds and MacLennan, 2005). Whilst the scattering coefficient sA is commonly used as a proxy for biomass, t/ha

values were also calculated to allow comparison with the diver based UVC estimates. The t/ha units are also more convenient and interpretable than the original units of m^2/ha (Boswell et al. 2007). This type of conversion has previously been undertaken for a mixed species communities by taking mean 'a' and 'b' values in the W-L formula for the species present (e.g. Wanzenböck et al. (2003); Boswell et al. (2007)), but we took this further by applying general values for 'a' and 'b' from Froese (2006). Whilst the imprecision from using this general length-weight formula is acknowledged, a strong positive correlation between our t/ha values and the m^2/ha sA values indicates the general viability in the calculated t/ha values for fish biomass.

4.5.2 Hydroacoustics and MPA assessment

To fully determine the effects of MPA placement on local fish populations, a Before, After, Control, Impact (BACI) design is necessary (Bernstein and Zalinski, 1983; Stewart-Oaten et al. 1986; Claudet et al. 2006), with the establishment of the park as the 'impact'. Underwood (1994) takes this further stating that more than a single control is necessary to reduce the likelihood of coincidental change. Ideally at least 2 sites with each combination of habitat and protection would be required to more fully determine protection effects (Guidetti, 2002). The choice of such independent control sites is, however, difficult in a heterogeneous environment and posed considerable logistic and financial constraints (Warwick, 1993; Halpern, 2003), but is recommended in future studies adopting a similar survey approach to our own. In determining the effects of an MPA, care must be taken so that protected area effects are not exaggerated or masked by other effects such as habitat (Claudet et al. 2008; Lester et al. 2009), as MPAs are often placed in particularly rich habitats (Caselle et al. 2015). Whilst it was not possible to survey the CPNP prior to its implementation with our hydroacoustic approach, we were able to provide two control sites that were open to fishing and containing the predominant habitat types found within the CPNP. The main combinations of habitat as well as exposure to fishing (protected *vs* not protected) were therefore accounted for in our survey design, however investigations into habitat effects were taken further by examining the fish associated with the reefs present within the CPNP separately instead of only the park as a whole. For an ideally balanced design we would have had transects from outside the park over similar reefs structures, however, this was not possible as none are present in the neighbouring areas. To account for this when we compared the standard CPNP transects with the controls, the reef-specific transects within the park were not included in the analyses.

The overall higher fish density and biomass across the CPNP is potentially a result of a "spillover" effect (McClanahan and Mangi, 2000; Roberts et al. 2001; Russ et al. 2004; Abesamis and Russ, 2005) occurring from significantly higher densities of fish associated with the reefs inside the park. In turn, it is possible that a spillover effect from the CPNP may have also increased fish density, biomass and size at the control sites. Although the whole park is effectively managed as a no-take area by the local community (Jones, 2011), there are unfortunately no data available on the levels of fishing at the control sites, although active fishing activity was observed at both locations during our surveys. The high density of small fish at PA may be due to high levels of size-selective fishing practices occurring here rather than, or in combination with, any habitat effects. This would have the effect of leaving a greater proportion of smaller fish in an area and also increasing the number of prey (a "prey release") following removal of the larger predators (Edgar et al. 2009; Sweeting et al. 2009; Edgar et al. 2014; Boaden and Kingsford 2015). This however requires further investigation using fisheries data from these areas, as well as more detailed hydroacoustic surveys with a temporal aspect to capture changes in fishing behaviours and fish densities.

4.5.3 Habitat effects

The relationship between marine fish species and their habitats is a key component in understanding their distributions (van der Kooij et al. 2011). Further, habitat complexity has long been known to have a positive effect on fish abundance (Luckhurst and Luckhurst 1978; Wilson et al. 2010; Komyakova et al. 2013; Graham and Nash 2013) and biomass (Grigg 1994). Our results agree with such findings that the most complex habitat, in this case the basalt reefs inside the CPNP, yield the highest fish density, biomass and mean size (Thesis Hypothesis 5a). Increased habitat complexity has been shown to have a strong positive effect on adult fish density and a weaker effect on recruit abundance (Almany 2004). Using hydroacoustics, Boswell et al. (2007) found significantly smaller fish over sandy habitats in comparison to more rugose habitats. The effect of habitat complexity may therefore be more pronounced with certain size classes of fish (Graham et al. 2006). Excluding the reef-specific transects, the highest fish density, biomass and size values were found within the CPNP area which is composed of a mix of the predominant habitats found at the control sites: heterogeneous sand/boulder/reef habitats. Between the control sites, greater numbers of fish (but not biomass) were present at PA compared to BS, a surprising result as BS contains a complex rocky habitat more likely to favour higher fish biomass similar to the rocky areas within the park. However,

this demonstrates how small fish do not contribute greatly to biomass levels at PA, despite relatively high densities (Wanzenböck et al. 2003). The higher fish density, biomass and mean fish size present over the reef specific transects could also be due to the generally shallower depths of these in comparison to the other sites. In other locations the overall and relative abundances of different trophic groups of fishes has been revealed at different depths (Friedlander and Parrish, 1998; Asher et al. 2017). The detailed analysis of fish distribution with depth was beyond the scope of the study, due to the large variation in along transect depths, and this is another reason why it was appropriate to separate these reef-specific sites out from the analysis of the CPNP vs control sites. In the examination of the CPNP vs control sites the mean water depths investigated along transects were however similar.

4.5.4 UVC surveys versus hydroacoustic surveys

Our density values differed significantly from those of the UVC surveys from the same reefs in the same year, although our biomass values did not (Thesis Hypotheses 3). Examining the size class distributions resultant from the two methods, this can be explained by the UVCs recording more fish in the smaller size classes than our hydroacoustic methods. Acoustics should not be biased in detecting these smaller fish when they are sufficiently separated from the reef matrix. It is, however likely that many of these small individuals are more cryptic and substrate-affiliated in nature than larger fishes (Ackerman et al. 2004). Our density and biomass values, are therefore likely to be conservative as smaller fish with closer associations with the seabed (within the "acoustic dead zone") will likely not have been counted by our hydroacoustic methods (Ona and Mitson, 1996; Zenone et al. 2017). Further, it is likely that this effect will have been more pronounced in areas of more complex habitat such as the reefs and boulders and if areas with overhangs and caves are present, then fish densities would certainly be underestimated.

Differences in fish density estimates could also be caused by potential differences in the precise locations of diver surveys on the reefs compared to the hydroacoustic transects. Differences in fish avoidance behaviour between the acoustic survey vessel and survey divers could also explain some differences in density estimates e.g. Draštík and Kubečka, 2005; Schmidt and Gassner, 2006. Finally, it is possible that differences in fish densities between the UVC and acoustic surveys may be caused by temporal variability. Both UVC and hydroacoustics were, however, conducted during daylight hours (avoiding crepuscular periods). It should however

be noted that both the hydroacoustic and UVC datasets represent snapshots in time and further interseasonal and interannual surveys would be of much merit. Little seasonal variation in the fish assemblages over the Cabo Pulmo reefs has however been noted (Alvarez-Filip et al. 2006).

Overall our acoustic survey campaign took one researcher and one boat operator a total of 8 days to survey the whole park twice as well as the two control sites and the final reef-specific surveys inside the CPNP. Hydroacoustics have the capacity to cover a greater area in a similar amount of time compared to UVC surveys. Furthermore, acoustics are not hampered by issues which can make areas inaccessible to SCUBA such as water clarity, strong currents or diver depth limits. Both hydroacoustic and UVC methods, however, can be hampered by adverse sea states (Knudsen, 2009). The start-up costs for the hydroacoustic equipment may be an impediment to their adoption for MPA evaluations, as we estimate they are approximately double that of an equivalent UVC SCUBA team (including training, certification and equipment). The UVC surveys in CPNP took 4 divers, 6 days to survey 12 reefs within the park which corresponds approximately to 0.1% of the total park area. UVC, however, can provide high-resolution species-specific information from which to detect subtler ecosystem shifts than changes in overall measures (e.g. density, biomass, size) (Hughes, 1994; Aburto-Oropeza et al. 2011). UVC surveys can also give additional information on MPA performance such as habitat health and invertebrate surveys which cannot be assessed through the hydroacoustic method we present here. Further, UVC will provide more detail on demersal species whilst hydroacoustics gives more information throughout the water column. We therefore conclude that is considerable merit in nesting UVC surveys within a hydroacoustic survey campaign, to provide higher resolution species-specific information in conjunction with the broader scale estimates of fish density, biomass and size (see also Murphy and Jenkins, 2010).

Our hydroacoustic surveys revealed important information on the nature of fish distributions inside, outside and amongst the differing habitats of the CPNP. This study highlights the importance of both protection and habitat in producing high fish density, biomass and mean sizes, emphasising the need to account for differences in habitat when designing coastal MPAs. Hydroacoustic surveys represent a valuable, non-invasive tool for the assessment of MPA fish populations, something that until now has been underutilised in MPA formation and management.

4.6 Appendices

4.6.1 Appendix 1

Table 4.5 Table of fish species recorded from the 2015 UVC surveys over the Reefs in the Cabo Pulmo National Park, with abundances, trophic group, mean sizes and species specific biomass.

Species	Abundance (# fish)	Mean size (cm)	(±S.E.M)	Biomass (tonne/ha)
Carnivores				
Anisotremus taeniatus	65	29.92	0.8	0.174
Arothron meleagris	77	20.71	0.53	0.023
Balistes polylepis	11	12.73	2.17	0.003
Bodianus diplotaenia	236	13.62	0.56	0.028
Canthigaster punctatissima	338	5		0.001
Chaetodon humeralis	10	20		0.027
Chanos chanos	74	80.27	1.15	8.833
Cirrhitichthys oxycephalus	1,039	5.13	0.03	0.002
Dasyatis dipterura	1	60		0.002
Decapterus muroadsi	100	10		0.011
Diodon holocanthus	42	21.07	0.81	0.018
Diodon hystrix	5	48	1.22	0.072
Epinephelus labriformis	70	24.14	0.64	0.031
Forcipiger flavissimus	1	15		0.001
Gymnothorax castaneus	11	103.82	9.74	0.061
Halichoeres chierchiae	6	14.17	1.54	0.001
Halichoeres dispilus	288	10.5	0.32	0.009
Halichoeres melanotis	4	17.5	2.5	0.001
Halichoeres nicholsi	1	25		0.003
Halichoeres notospilus	3	25		0.003
Johnrandallia nigrirostris	66	13.11	0.5	0.007
Lutjanus viridis	77	22.6	0.33	0.029
Mulloidichthys dentatus	111	22.66	0.48	0.044
Muraena lentiginosa	1	35		0.001
Novaculichthys taeniourus	1	25		0.002
Ostracion meleagris	9	10.56	1.3	0.001
Plagiotremus azaleus	133	6.88	0.21	0.0003
Pomacanthus zonipectus	1	30		0.008
Pseudobalistes naufragium	4	52.5	5.2	0.058
Rypticus bicolor	2	7.5	2.5	0.0001
Serranus psittacinus	36	12.22	0.81	>0.0001
Sufflamen verres	124	16.33	0.78	0.025

Thalassoma grammaticum	231	7.45	0.35	0.005
Thalassoma lucasanum	2,520	6.81	0.17	0.144
Zanclus cornutus	52	14.52	0.63	0.015
Herbivores				
Acanthurus nigricans	16	18.13	1.43	0.007
Acanthurus xanthopterus	57	28.77	1.06	0.072
Holacanthus clarionensis	1	10		0.0003
Holacanthus passer	135	17.11	0.63	0.038
Kyphosus analogus	8	37.5	0.94	0.013
Kyphosus elegans	1	30		0.005
Microspathodon dorsalis	6	12.5	3.59	0.003
Nicholsina denticulata	1	20		0.001
Ophioblennius steindachneri	67	13.36	0.59	0.004
Prionurus punctatus	415	25.1	0.26	0.27
Scarus compressus	4	38.75	9.44	0.031
Scarus ghobban	23	45	3.16	0.058
Scarus perrico	3	31.67	6.01	0.011
Scarus rubroviolaceus	26	43.08	3.27	0.066
Stegastes flavilatus	366	7.1	0.13	0.004
Stegastes rectifraenum	630	8.02	0.11	0.01
Piscivores				
Aulostomus chinensis	1	10		>0.0001
Carangoides orthogrammus	8	35.63	1.48	0.032
Caranx caballus	10	31.5	1.5	0.063
Caranx sexfasciatus	50	45		1.217
Cephalopholis panamensis	44	19.66	1.39	0.013
Cirrhithus rivulatus	4	22.5	2.5	0.005
Fistularia commersonii	18	96.11	9.35	0.045
Gnathanodon speciosus	2	35		0.02
Hoplopagrus guentherii	4	43.75	5.15	0.043
Lutjanus argentiventris	20	50.5	2.11	0.5
Lutjanus novemfasciatus	4	65	5	0.168
Mycteroperca rosacea	121	48.55	1.15	0.357
Zooplanktivores				
Abudefduf troschelii	16	18.44	1.09	0.02
Chromis atrilobata	2,039	6.32	0.05	0.009
Myripristis leiognathus	95	9.26	0.64	0.021
Paranthias colonus	917	20.75	0.23	0.108

4.6.2 Appendix 2

Data on fish abundance and biomass sA is plotted geographically in Figure 4.7 and Figure 4.8, below.
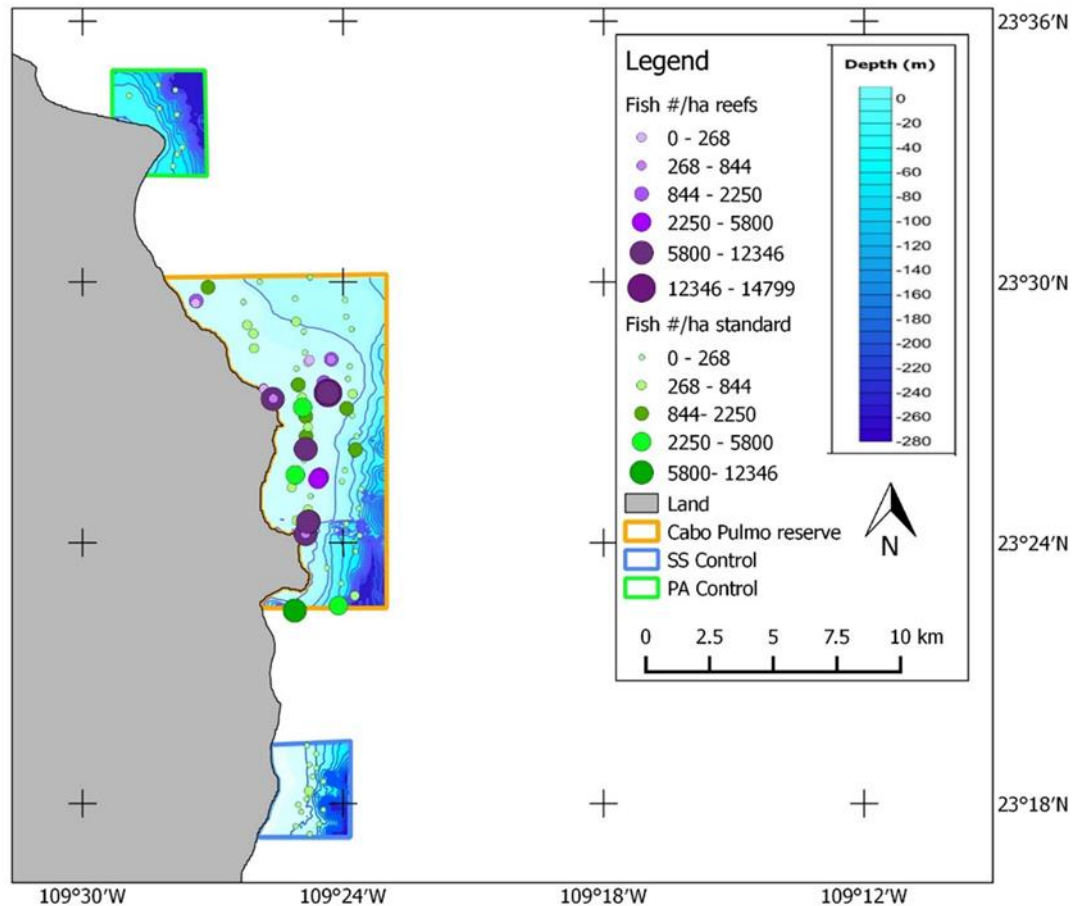
Figure 4.7 Fish abundance data (number of fish per hectare) plotted geographically as midpoints along transects at the Cabo Pulmo National Park and Control areas. Standard transect means are shown by green bubble values whilst reef-specific transects in purple. Bathymetry is also indicated. Coordinates are in WGS 84.
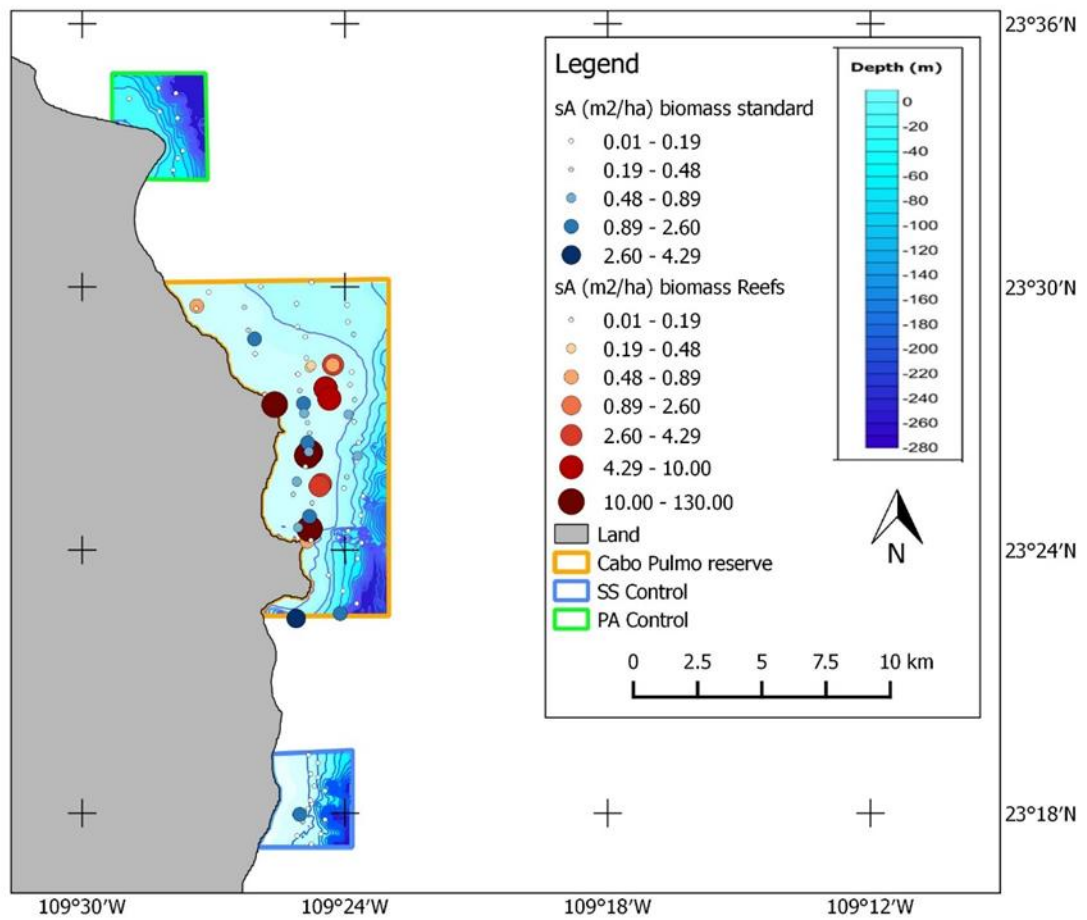
Figure 4.8 Fish sA data (biomass proxy in m² per hectare) plotted geographically as midpoints along transects at the Cabo Pulmo National Park and Control areas. Standard transect means are shown by blue bubble values whilst reef-specific transects in orange. Bathymetry is also shown. Coordinates are in WGS 84.

4.7   References

Abesamis, R.A. and Russ, G.R., 2005. Density-dependent spillover from a marine reserve: Long-term evidence. *Ecological applications*, *15*(5), pp.1798-1812.

Aburto-Oropeza, O., Ezcurra, E., Moxley, J., Sánchez-Rodríguez, A., Mascarenas-Osorio, I., Sánchez-Ortiz, C., Erisman, B. and Ricketts, T., 2015. A framework to assess the health of rocky reefs linking geomorphology, community assemblage, and fish biomass. *Ecological Indicators*, *52*, pp.353-361.

Aburto-Oropeza, O., Erisman, B., Galland, G.R., Mascareñas-Osorio, I., Sala, E. and Ezcurra, E., 2011. Large recovery of fish biomass in a no-take marine reserve. *PLoS One*, *6*(8), p.e23601.

Ackerman, J.L., Bellwood, D.R. and Brown, J.H., 2004. The contribution of small individuals to density-body size relationships: examination of energetic equivalence in reef fishes. *Oecologia*, *139*(4), pp.568-571.

Aglen, A. 1989. Empirical results on precision-effort relationships for acoustic surveys. *ICES Journal of Marine Science*: 1989/B:30, 28 p

Allison, G. W., Lubchenco, J. & Carr, M. H. Marine reserves are necessary but not sufficient for marine conservation. *Ecological Applications.* **8,** 79–92 (1998).

Almany, G.R., 2004. Differential effects of habitat complexity, predators and competitors on abundance of juvenile and adult coral reef fishes. *Oecologia*, *141*(1), pp.105-113.

Alvarez-Filip, L., Reyes-Bonilla, H. and Calderon-Aguilera, L.E., 2006. Community structure of fishes in Cabo Pulmo reef, Gulf of California. *Marine Ecology*, *27*(3), pp.253-262.

Asher, J.M., Williams, I. and Harvey, E., 2017. Mesophotic Depth Gradients Impact Reef Fish Assemblage Composition and Functional Group Partitioning in the Main Hawaiian Islands. *Frontiers in Marine Science*, 4, p.98.

Balk, H. and Lindem, T. 2006. Sonar 4, Sonar 5, Sonar 6 e Post-processing Systems. Operator Manual. Oslo: Lindem Data Acquisition.

Bernstein, B.B. and Zalinski, J., 1983. An optimum sampling design and power tests for environmental biologists. *Journal of Environmental Management*, *16*(1), pp.35-43.

Beverton, R.J. and Holt, S.J., 1957. *On the dynamics of exploited fish populations* (Vol. 11). Springer Science and Business Media.

Boaden, A.E. and Kingsford, M., 2015. Predators drive community structure in coral reef fish assemblages. *Ecosphere*, *6*(4), pp.1-33.

Boersma, P. D. & Parrish, J. K. Limiting abuse: Marine protected areas, a limited solution. *Ecological Economics.* **31,** 287–304 (1999).

Boswell, K.M., Wells, R.J., Cowan Jr, J.H. and Wilson, C.A., 2010. Biomass, density, and size distributions of fishes associated with a large-scale artificial reef complex in the Gulf of Mexico. *Bulletin of Marine Science*, *86*(4), pp.879-889.

Boswell, K.M., Wilson, M.P. and Wilson, C.A., 2007. Hydroacoustics as a tool for assessing fish biomass and size distribution associated with discrete shallow water estuarine habitats in Louisiana. *Estuaries and Coasts*, *30*(4), pp.607-617.

Caselle, J. E., Rassweiler, A., Hamilton, S. L. & Warner, R. R. Recovery trajectories of kelp forest animals are rapid yet spatially variable across a network of temperate marine protected areas. *Scientific Reports*. **5,** 14102 (2015).

Claudet, J., Pelletier, D., Jouvenel, J.Y., Bachet, F. and Galzin, R., 2006. Assessing the effects of marine protected area (MPA) on a reef fish assemblage in a northwestern Mediterranean marine reserve: Identifying community-based indicators. *Biological Conservation*, *130*(3), pp.349-369.

Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., García-Charton, J.A., Pérez-Ruzafa, Á., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F. and Culioli, J.M., 2008. Marine reserves: size and age do matter. *Ecology Letters*, *11*(5), pp.481-489.

Coll, C., de Morais, L.T., Laë, R., Lebourges-Dhaussy, A., Simier, M., Guillard, J., Josse, E., Ecoutin, J.M., Albaret, J.J., Raffray, J. and Kantoussan, J., 2007. Use and limits of three methods for assessing fish size spectra and fish abundance in two tropical man-made lakes. *Fisheries Research*, *83*(2), pp.306-318.

Crain, C.M., Halpern, B.S., Beck, M.W. and Kappel, C.V., 2009. Understanding and managing human threats to the coastal marine environment. *Annals of the New York Academy of Sciences*, *1162*(1), pp.39-62.

Diaz, R.J., Solan, M. and Valente, R.M., 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management*, *73*(3), pp.165-181.

Draštík, V., Kubečka, J., Čech, M., Frouzová, J., Říha, M., Juza, T., Tušer, M., Jarolím, O., Prchalová, M., Peterka, J. and Vašek, M., 2009. Hydroacoustic estimates of fish stocks in temperate reservoirs: day or night surveys?. *Aquatic Living Resources*, *22*(1), pp.69-77.

Draštík, V. and Kubečka, J., 2005. Fish avoidance of acoustic survey boat in shallow waters. *Fisheries Research*, *72*(2), pp.219-228.

Edgar, G.J., Stuart-Smith, R.D., Willis, T.J., Kininmonth, S., Baker, S.C., Banks, S., Barrett, N.S., Becerro, M.A., Bernard, A.T., Berkhout, J. and Buxton, C.D., 2014. Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*(7487), pp.216-220.

Edgar, G.J., Barrett, N.S. and Stuart-Smith, R.D., 2009. Exploited reefs protected from fishing transform over decades into conservation features otherwise absent from seascapes. *Ecological Applications*, *19*(8), pp.1967-1974.

Egerton, J.P., Johnson, A.F., Le Vay, L., McCoy, C.M., Semmens, B.X., Heppell, S.A. and Turner, J.R., 2017. Hydroacoustics for the discovery and quantification of Nassau grouper (*Epinephelus striatus*) spawning aggregations. *Coral Reefs*, *36*(2), pp.589-600.

Emmrich, M., Winfield, I.J., Guillard, J., Rustadbakken, A., Verges, C., Volta, P., Jeppesen, E., Lauridsen, T.L., Brucet, S., Holmgren, K. and Argillier, C., 2012. Strong correspondence between gillnet catch per unit effort and hydroacoustically derived fish biomass in stratified lakes. *Freshwater Biology*, *57*(12), pp.2436-2448.

Erisman, B.E., Galland, G.R., Mascarenas, I., Moxley, J., Walker, H.J., Aburto-Oropeza, O., Hastings, P.A. and Ezcurra, E., 2011. List of coastal fishes of Islas Marías archipelago, Mexico, with comments on taxonomic composition, biogeography, and abundance. *Zootaxa*, *2985*, pp.26-40.

Foote, K. G., Knudsen, H. P., Vestnes, G., MacLennan, D. N., and Simmonds. E. J. 1987. Calibration of acoustic instruments for fish density estimation: a practical guide. Cooperative Research Report. *ICES Journal of Marine Science, 144*, 69 pp

Friedlander, A.M., Brown, E.K., Jokiel, P.L., Smith, W.R. and Rodgers, K.S., 2003. Effects of habitat, wave exposure, and marine protected area status on coral reef fish assemblages in the Hawaiian archipelago. *Coral Reefs*, *22*(3), pp.291-305.

Friedlander, A.M., Brown, E.K. and Monaco, M.E., 2007. Coupling ecology and GIS to evaluate efficacy of marine protected areas in Hawaii. *Ecological Applications*, *17*(3), pp.715-730.

Friedlander, A.M. and Parrish, J.D., 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *Journal of Experimental Marine Biology and Ecology*, 224(1), pp.1-30.

Froese, R., 2006. Cube law, condition factor and weight–length relationships: history, meta-analysis and recommendations. *Journal of Applied Ichthyology*, *22*(4), pp.241-253.

Frouzova, J., Kubecka, J., Balk, H. and Frouz, J., 2005. Target strength of some European fish species and its dependence on fish body parameters. *Fisheries Research*, *75*(1), pp.86-96.

Game, E.T., Grantham, H.S., Hobday, A.J., Pressey, R.L., Lombard, A.T., Beckley, L.E., Gjerde, K., Bustamante, R., Possingham, H.P. and Richardson, A.J., 2009. Pelagic protected areas: the missing dimension in ocean conservation. *Trends in Ecology & Evolution*, 24(7), pp.360-369.

García-Charton, J.A., Pérez-Ruzafa, A., Marcos, C., Claudet, J., Badalamenti, F., Benedetti-Cecchi, L., Falcón, J.M., Milazzo, M., Schembri, P.J., Stobart, B. and Vandeperre, F., 2008. Effectiveness of European Atlanto-Mediterranean MPAs: Do they accomplish the expected effects on populations, communities and ecosystems?. *Journal for Nature Conservation,* 16(4), pp.193-221.

Graham, N.A., Wilson, S.K., Jennings, S., Polunin, N.V., Bijoux, J.P. and Robinson, J., 2006. Dynamic fragility of oceanic coral reef ecosystems. *Proceedings of the National Academy of Sciences*, *103*(22), pp.8425-8429.

Graham, N.A.J. and Nash, K.L., 2013. The importance of structural complexity in coral reef ecosystems. *Coral Reefs*, *32*(2), pp.315-326.

Grigg, R.W., 1994. Effects of sewage discharge, fishing pressure and habitat complexity on coral ecosystems and reef fishes in Hawaii. *Marine Ecology Progress Series*, pp.25-34.

Grober-Dunsmore, R., Wooninck, L., Field, J., Ainsworth, C., Beets, J., Berkeley, S., Bohnsack, J., Boulon, R., Brodeur, R., Brodziak, J. and Crowder, L., 2008. Vertical zoning in marine protected areas: ecological considerations for balancing pelagic fishing with conservation of benthic communities. *Fisheries*, 33(12), pp.598-610.

Guidetti, P. The importance of experimental design in detecting the effects of protection measures on fish in Mediterranean MPAs. *Aquatic Conservation Marine and Freshwater Ecosystems*. **12,** 619–634 (2002).

Halpern, B.S., 2003. The impact of marine reserves: do reserves work and does reserve size matter?. *Ecological Applications*, pp.S117-S137.

Harmelin-Vivien, M.L., Harmelin, J.G., Chauvet, C., Duval, C., Galzin, R., Lejeune, P., Barnabé, G., Blanc, F., Chevalier, R., Duclerc, J. and Lasserre, G., 1985. Evaluation visuelle des peuplements et populations de poissons méthodes et problèmes. *Revue d Ecologie.* Vol 40

Havard, L., Brigand, L. and Cariño, M., 2015. Stakeholder participation in decision-making processes for marine and coastal protected areas: Case studies of the south-western Gulf of California, Mexico. *Ocean & Coastal Management*, 116, pp.116-131.

Hill, N.A., Barrett, N., Lawrence, E., Hulls, J., Dambacher, J.M., Nichol, S., Williams, A. and Hayes, K.R., 2014. Quantifying fish assemblages in large, offshore marine protected areas: an Australian case study. *PloS One*, *9*(10), p.e110831.

Hughes, T.P., 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science-AAAS-Weekly Paper Edition*, *265*(5178), pp.1547-1551.

Johnson, A.F., Jenkins, S.R., Hiddink, J.G. and Hinz, H., 2013. Linking temperate demersal fish species to habitat: scales, patterns and future directions. *Fish and Fisheries*, *14*(3), pp.256-280.

Jones, N., 2011. Little Mexican reserve boasts big recovery. *Nature*. Available at: http://www.nature.com/doifinder/10.1038/news.2011.479

Jones, D.T., Wilson, C.D., De Robertis, A., Rooper, C.N., Weber, T.C. and Butler, J.L., 2012. Evaluation of rockfish abundance in untrawlable habitat: combining acoustic and complementary sampling tools. *Fishery Bulletin*, 110(3), pp.332-343.

Knudsen, H. P. Long-term evaluation of scientific-echosounder performance. *ICES Journal of Marine Science*, **66,** 1335–1340 (2009).

Komyakova, V., Munday, P.L. and Jones, G.P., 2013. Relative importance of coral cover, habitat complexity and diversity in determining the structure of reef fish communities. *PLoS One*, *8*(12), p.e83178.

van der Kooij, J., Kupschus, S. and Scott, B.E., 2011. Delineating the habitat of demersal fish assemblages with acoustic seabed technologies. *ICES Journal of Marine Science*, *68*(9), pp.1973-1985.

Kracker, L., 2007. Hydroacoustic surveys: A non-destructive approach to monitoring fish distributions at National Marine Sanctuaries. NOAA Technical Memorandum NOS NCCOS 66

Lester, S.E., Halpern, B.S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B.I., Gaines, S.D., Airamé, S. and Warner, R.R., 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, *384*, pp.33-46.

Love, R.H., 1971. Measurements of fish target strength: a review. *Fish. Bull*, *69*(4), pp.703-715.

Love, R.H. 1977. Target strength of an individual fish at any aspect. *The Journal of the Acoustical Society of America*, *62*(6), p.1397.

Luckhurst, B.E. and Luckhurst, K., 1978. Analysis of the influence of substrate variables on coral reef fish communities. *Marine Biology*, *49*(4), pp.317-323.

Mackinson, S., Freeman, S., Flatt, R. and Meadows, B., 2004. Improved acoustic surveys that save time and money: integrating fisheries and ground-discrimination acoustic technologies. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.129-140.

MacNeil, M.A., Tyler, E.H., Fonnesbeck, C.J., Rushton, S.P., Polunin, N.V. and Conroy, M.J., 2008. Accounting for detectability in reef-fish biodiversity estimates. *Marine Ecology Progress Series*, *367*, pp.249-260.

Mascarenas-Osorio, I., Erisman, B., Moxley, J., Balart, E.F. and Aburto-Oropeza, O., 2011. Checklist of conspicuous reef fishes of the Bahía de los Ángeles region, Baja California Norte, Mexico, with comments on abundance and ecological biogeography. *Zootaxa*, *2922*, pp.60-68.

McClanahan, T.R. and Mangi, S., 2000. Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. *Ecological Applications*, *10*(6), pp.1792-1805.

Miller, K.I. and Russ, G.R., 2014. Studies of no-take marine reserves: methods for differentiating reserve and habitat effects. *Ocean and Coastal Management*, *96*, pp.51-60.

Murphy, H.M. and Jenkins, G.P., 2010. Observational methods used in marine spatial monitoring of fishes and associated habitats: a review. *Marine and Freshwater Research*, *61*(2), pp.236-252.

Nagelkerken, I., Grol, M.G. and Mumby, P.J., 2012. Effects of marine reserves versus nursery habitat availability on structure of reef fish communities. *PLoS One*, *7*(6), p.e36906.

Nielsen, J.R. and Lundgren, B., 1999. Hydroacoustic *ex situ* target strength measurements on juvenile cod (Gadus morhua L.). ***ICES Journal of Marine Science***, *56*(5), pp.627-639.

Ona, E. and Mitson, R.B., 1996. Acoustic sampling and signal processing near the seabed: the deadzone revisited. ***ICES Journal of Marine Science***, *53*(4), pp.677-690.

Ortiz, D.M. and Tissot, B.N., 2008. Ontogenetic patterns of habitat use by reef-fish in a Marine Protected Area network: a multi-scaled remote sensing and *in situ* approach. ***Marine Ecology Progress Series***, *365*, pp.217-232.

Parker-Stetter, S.L. Rudstam, L.G., Sullivan, P.J., Warner, D.M. 2009. Standard operating procedures for fisheries acoustic surveys in the Great Lakes. Great Lakes Fisheries Commission Special Publication 09-01

Pollom, R.A. and Rose, G.A., 2015. Size-based hydroacoustic measures of within-season fish abundance in a boreal freshwater ecosystem. ***PloS One***, *10*(4), p.e0124799.

Polunin, N., Bloomfield, H., Sweeting, C. and McCandless, D., 2009. Developing indicators of MPA effectiveness: finfish abundance and diversity in a Yorkshire prohibited trawl area. *Report to MFA, London*.

Reid, D.G. 2000. ICES Cooperative Research Report No. 238. 2000. Report on Echo Trace Classification. David G. Reid (ed). No. 286. pp 183.

Reyes-Bonilla, H. and Calderon-Aguilera, L.E. 1999. Population Density, Distribution and Consumption Rates of Three Corallivores at Cabo Pulmo Reef, Gulf of California, Mexico. ***Marine Ecology***, *20*(3–4), pp.347–357.

Riegl, B.M., Halfar, J., Purkis, S.J. and Godinez-Orta, L., 2007. Sedimentary facies of the eastern Pacific's northernmost reef-like setting (Cabo Pulmo, Mexico). *Marine* ***Geology***, *236*(1), pp.61-77.

Roberts, C.M., Bohnsack, J.A., Gell, F., Hawkins, J.P. and Goodridge, R., 2001. Effects of marine reserves on adjacent fisheries. ***Science***, *294*(5548), pp.1920-1923.

Rudershausen, P.J., Mitchell, W.A., Buckel, J.A., Williams, E.H. and Hazen, E., 2010. Developing a two-step fishery-independent design to estimate the relative abundance of deepwater reef fish: application to a marine protected area off the southeastern United States coast. ***Fisheries Research***, *105*(3), pp.254-260

Rudstam, L.G., Parker-Stetter, S.L., Sullivan, P.J. and Warner, D.M., 2009. Towards a standard operating procedure for fishery acoustic surveys in the Laurentian Great Lakes, North America. ***ICES Journal of Marine Science***, *66*(6), pp.1391-1397.

Russ, G.R., Alcala, A.C., Maypa, A.P., Calumpong, H.P. and White, A.T., 2004. Marine reserve benefits local fisheries. ***Ecological Applications***, *14*(2), pp.597-606..

Sale, P.F., Cowen, R.K., Danilowicz, B.S., Jones, G.P., Kritzer, J.P., Lindeman, K.C., Planes, S., Polunin, N.V., Russ, G.R., Sadovy, Y.J. and Steneck, R.S., 2005. Critical science gaps impede use of no-take fishery reserves. ***Trends in Ecology & Evolution***, 20(2), pp.74-80.

Sawada, K., Furusawa. M., Williamson. N.J., 1993. Conditions for the precise measurement of fish target strength in situ. *Journal of the Marine Acoustical Society*. *20*; 15–21.

Schmidt, M.B. and Gassner, H., 2006. Influence of scuba divers on the avoidance reaction of a dense vendace (*Coregonus albula* L.) population monitored by hydroacoustics. *Fisheries Research*, *82*(1), pp.131-139.

Sciberras, M., Jenkins, S.R., Mant, R., Kaiser, M.J., Hawkins, S.J. and Pullin, A.S., 2015. Evaluating the relative conservation value of fully and partially protected marine areas. *Fish and Fisheries*, *16*(1), pp.58-77.

Simmonds, J. and MacLennan, D.N., 2005. *Fisheries acoustics: theory and practice*. John Wiley and Sons.

Starr, R.M., Wendt, D.E., Barnes, C.L., Marks, C.I., Malone, D., Waltz, G., Schmidt, K.T., Chiu, J., Launer, A.L., Hall, N.C. and Yochum, N., 2015. Variation in responses of fishes across multiple reserves within a network of marine protected areas in temperate waters. *PloS One*, *10*(3), p.e0118502.

Stewart-Oaten, A., Murdoch, W.W. and Parker, K.R., 1986. Environmental impact assessment:" pseudoreplication" in time?. *Ecology*, *67*(4), pp.929-940.

Stewart, G.B., Kaiser, M.J., Côté, I.M., Halpern, B.S., Lester, S.E., Bayliss, H.R. and Pullin, A.S., 2009. Temperate marine reserves: global ecological effects and guidelines for future networks. *Conservation Letters*, *2*(6), pp.243-253.

Sweeting, C.J., Badalamenti, F., D'Anna, G., Pipitone, C. and Polunin, N.V.C., 2009. Steeper biomass spectra of demersal fish communities after trawler exclusion in Sicily. *ICES Journal of Marine Science*, *66*(1), pp.195-202.

Thiriet, P.D., Di Franco, A., Cheminée, A., Guidetti, P., Bianchimani, O., Basthard-Bogain, S., Cottalorda, J.M., Arceo, H., Moranta, J., Lejeune, P. and Francour, P., 2016. Abundance and Diversity of Crypto-and Necto-Benthic Coastal Fish Are Higher in Marine Forests than in Structurally Less Complex Macroalgal Assemblages. *PloS One*, 11(10), p.e0164121.

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C., 2011. Underwater acoustics for ecosystem-based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, *442*, pp.285-301.

Underwood, A.J., 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications*, *4*(1), pp.3-15.

Verutes, G.M., Huang, C., Estrella, R.R. and Loyd, K., 2014. Exploring scenarios of light pollution from coastal development reaching sea turtle nesting beaches near Cabo Pulmo, Mexico. *Global Ecology and Conservation*, *2*, pp.170-180.

Wanzenböck, J., Mehner, T., Schulz, M., Gassner, H. and Winfield, I.J., 2003. Quality assurance of hydroacoustic surveys: the repeatability of fish-abundance and biomass estimates in lakes within and between hydroacoustic systems. *ICES Journal of Marine Science*, *60*(3), pp.486-492.

Warwick, R.M., 1993. Environmental impact studies on marine communities: pragmatical considerations. *Austral Ecology*, *18*(1), pp.63-80.

Wheeland, L.J. and Rose, G.A., 2015. Acoustic measures of lake community size spectra. ***Canadian Journal of Fisheries and Aquatic Sciences***, *73*(4), pp.557-564.

Wilson, S.K., Fisher, R., Pratchett, M.S., Graham, N.A.J., Dulvy, N.K., Turner, R.A., Cakacaka, A. and Polunin, N.V.C., 2010. Habitat degradation and fishing effects on the size structure of coral reef fish communities. ***Ecological Applications***, *20*(2), pp.442-451.

Winfield, I.J., Emmrich, M., Guillard, J., Mehner, T. and Rustadbakken, A., 2011. WISER Deliverable 3.4-3: Guidelines for standardisation of hydroacoustic methods. *WISER Project Newsletter*, *2*, pp.4-4.

Yule, D.L., Adams, J.V., Hrabik, T.R., Vinson, M.R., Woiak, Z. and Ahrenstorff, T.D., 2013. Use of classification trees to apportion single echo detections to species: application to the pelagic fish community of Lake Superior. ***Fisheries Research***, *140*, pp.123-132.

Yurista, P.M., Yule, D.L., Balge, M., VanAlstine, J.D., Thompson, J.A., Gamble, A.E., Hrabik, T.R., Kelly, J.R., Stockwell, J.D. and Vinson, M.R., 2014. A new look at the Lake Superior biomass size spectrum. ***Canadian Journal of Fisheries and Aquatic Sciences***, *71*(9), pp.1324-1333.

Zenone, A.M., Burkepile, D.E. and Boswell, K.M., 2017. A comparison of diver vs. acoustic methodologies for surveying fishes in a shallow water coral reef ecosystem. ***Fisheries Research***, *189*, pp.62-66.

5 Chapter 5. Fish association with shallow offshore habitats in the Qatari waters of the Arabian Gulf

5.1 Abstract

In order to implement spatial fisheries management in the Arabian Gulf, a better understanding of the distribution of fish in relation to benthic habitats is required. To facilitate this, hydroacoustic fish surveys were conducted over oyster bed/reef ("shallow") and surrounding soft sediment ("deep") habitats in the offshore central Gulf, within Qatari waters. Transects at 'shallow' sites had significantly higher mean fish density and biomass. Mean target strength of individual fish was also significantly greater at 'shallow' sites. Fish positions in the water column were examined and overall there was a closer association with the seabed at the 'shallow' sites. Larger fish were found significantly closer to the seabed than smaller fish across all sites, but more so at 'shallow' sites than at 'deep' sites. Acoustic return from the seabed was extracted to provide information on the habitat type both using 'Sonar5 (ver 6.0.4)' and 'Visual Habitat' software. The different site categories ('shallow' vs 'deep') were significantly different for all the measures of acoustic habitat. Fish density was significantly related to 'Visual Habitat' data, more so than depth alone. Our results show that fish distribution in the offshore Gulf is associated with complex, shallow oyster bed/reef habitats, and this is particularly the case for larger demersal fish that are commercially exploited. The ability to characterise benthic habitats from acoustic fish survey data shows promise, with important time-saving implications for the monitoring of marine environments and developing a spatial approach to fisheries management. This may include the identification of habitats with a relatively high density of larger fish for inclusion in candidate marine protected areas.

5.2 Introduction

Fish are a vital source of protein throughout the world and the demand for fish resources continues to increase. This is also the case in the Arabian Gulf (hereafter referred to as 'the Gulf'), where rapid coastal development has been accompanied by high human population growth (Feidi, 1998). This population growth will continue to increase pressures on fish stocks, especially on demersal high value species that are already reported as fully or overexploited in the area (De Young, 2006). This overfishing (Siddeek et al. 1999) has already resulted in a rapid decline in the health and sustainability of the Gulf ecosystem (Sheppard et al. 2010; Sale et al. 2011; Feary et al. 2011). Effective management of fish resources is therefore necessary in order to ensure that any overfishing is reduced and sustainability prevails (Pauly et al. 2002). A shift towards resource management that is ecosystem-based with long-term perspectives is urgently needed in the Gulf (Khan, 2007). From a fisheries science perspective, one step towards effective management is to develop an understanding of fish distribution and fish-habitat linkages as a component of Ecosystem Based Fisheries Management (EBFM) (Larkin, 1996). Relating marine fish with specific habitats is however a difficult task obscured by uncertainty due to the variety of habitats used over fish lifetimes, large variations in fish density and complex spatial heterogeneity in habitats (Rose, 2000; Minns and Moore, 2003; Anderson, 2008). Nevertheless, hydroacoustics have shown that seabed substratum is one of the most important components determining the spatial ecology of demersal fish (Ellis et al. 2000; McConnaughey and Syrjala, 2009; Moore et al. 2009; van der Kooij et al. 2011) and also with pelagic species (Maravelias et al. 2006).

Benthic habitat is primarily determined by substrate type (Kostylev et al. 2001) and throughout this manuscript we use the term 'habitat' to describe what others may term 'substrate' (Diaz et al. 2004). The most widespread habitats offshore in the Gulf are muddy and sandy substrata (Sheppard et al. 2010; Feary et al. 2011), however these are interspersed by shallower limestone outcrops (Riegl, 1999). These shallower outcrops (locally known as 'hairãt') provide a hard substrate that is typically colonised by benthic epifauna including oyster beds and corals (Riegl, 1999; Sheppard et al. 2010; Smyth et al. 2016). There are no true coral reefs in the Gulf (Sale et al. 2011), rather corals form more of a veneer over the hard substrates present (Riegl, 1999; Sheppard et al. 2010; Feary et al. 2011; Sale et al. 2011). When hard substrates do host coral communities, these areas provide habitat to a relatively abundant and diverse fish community in the Gulf (Feary et al. 2011).

Qatari fisheries are artisanal in terms of methods but are active on a large scale (Al-Abdulrazzak, 2013). Fishing in Qatari waters occurs almost entirely on the eastern side of the peninsula in offshore waters of the central Gulf, mostly less than 50m depth (Al-Ansi and Priede, 1996). Industrial trawling was banned in Qatari waters in 1992 and since then the demersal catch has increased through the use of gill nets, hook and line and fish traps (gargoor) (Al-Ansi and Priede, 1996, Siddeek et al. 1999). Landings of demersal species represented around 71% of the total catch in Qatar in 1992 and 1993 (Siddeek et al. 1999). The demersal fish most commonly targeted by the Qatar trap fishery are *Lethrinus* and *Epinephelus* spp, which together account for around 29% of the annual total catch in Qatar (Stamatopoulos and Abdallah, 2016). Demersal fishing effort tends to be focused on traditional offshore fishing grounds which include the shallow 'hairãt' habitats, which are considered highly productive and support high benthic biodiversity (Smyth et al. 2016). Such characteristics would justify the inclusion of these habitats in protected areas for both biodiversity conservation and spatial management of fish stocks. However, to date there is limited evidence to confirm their role as essential fish habitat (EFH). Whilst there has been some historic effort in determining the distribution of fish in the region via scientific trawling (e.g. Sivasubramaniam and Ibrahim, 1982) this has largely been confined to the softer sediments, due to safety issues and potential damage to both fishing gear and to the reefs themselves. The hydroacoustic method however allows a comparable methodology over the different habitats. Additional advantages of the methodology include rapid acquisition and retention of raw data and any size selectivity of fishing gear is removed (Trenkel et al. 2011).

Hydroacoustics can be the most efficient remote sensing tool for mapping and monitoring the subsurface oceans over large areas (Anderson et al. 2008). To further increase efficiencies, the same hydroacoustic fish data can also be processed to give information on habitat type with time and cost saving implications (Freeman et al. 2004; Mackinson et al. 2004). The coverage of the data is also likely to be greater than that of traditional point sampling techniques for habitat mapping (Freitas et al. 2008). There are a number of bespoke acoustic ground discrimination systems (AGDS) used for habitat mapping (e.g. RoxAnn, QTC-View, EchoPlus) (Brown et al. 2011) which categorise the acoustic responses from the seabed based on roughness and hardness (Foster-Smith and Sotheran, 2003). Recently, Biosonics Inc have released Visual Habitat (VH) software that can be used in conjunction with their DTX echosounders, which we examine for discriminating between the different habitat types present

within the survey area. Additionally we examine how acoustic reflection parameters from the seabed extracted from Sonar5 (ver 6.0.4) (Balk and Lindem, 2006) compare with the habitat data given by VH. Hydroacoustic data were also processed to investigate fish height in the water column over the different habitats present. Such data is often examined to help classify fish echoes into species groups (e.g. Parker-Stetter et al. 2009), and to examine diel vertical migration (DVM) (e.g. Hrabik et al. 2006; Jensen et al. 2011). There has however been little use of such data to examine fish utilisation of habitat. We investigate how this data can be used, in addition to fish size, to further highlight any effects of benthic habitat on the vertical distribution of fish between study sites.

In this study we use hydroacoustics to help understand fish distribution in Qatari waters of the central Gulf through examining fish-habitat linkages in order to test the potential role of shallow oyster beds/reefs as fish habitat (Thesis Hypothesis 5a and b). Through testing the hypothesis that these areas have a greater density of fish and a larger mean size (Thesis Hypothesis 5a), we aim to provide evidence that can inform future planning and aid the development of appropriate Ecosystem Based Fisheries Management (EBFM) in the region.

5.3 Methods

5.3.1 Study sites

Acoustic surveys were performed from 3rd to 7th of May 2015, from a 26ft motor boat working alongside the Qatar University research vessel RV Janan which was used for other aspects of the overall study (towed camera, diving, and fishing) and accommodation. Sites were chosen through examination of bathymetric charts and local knowledge. All sites are shown in Fig. 1 and locations, depths and groundtruthed habitat type are given in Table 5.1.

'Shallow' sites: These sites aimed to target the raised limestone mounds that have a patchy distribution amongst the surrounding deeper waters with muddier sediments. These mounds are mainly located in water depths of 10-20m and are hereon referred to as 'shallow' sites. They have more consolidated coarse and rugose substrate, and are typically colonised by oyster bed or mixed reefs communities (Smyth et al 2016). Of the sites included in this study, there is most live coral at the site of Halul Island (site S6), where five species have been recorded in recent surveys (Sheppard et al. 2010).

'Deep' sites: Sites located in the deeper waters surrounding the raised mounds are referred to as 'deep' sites. These are in water depths of circa 25-40m comprising finer and more mobile sediments of sand and mud.
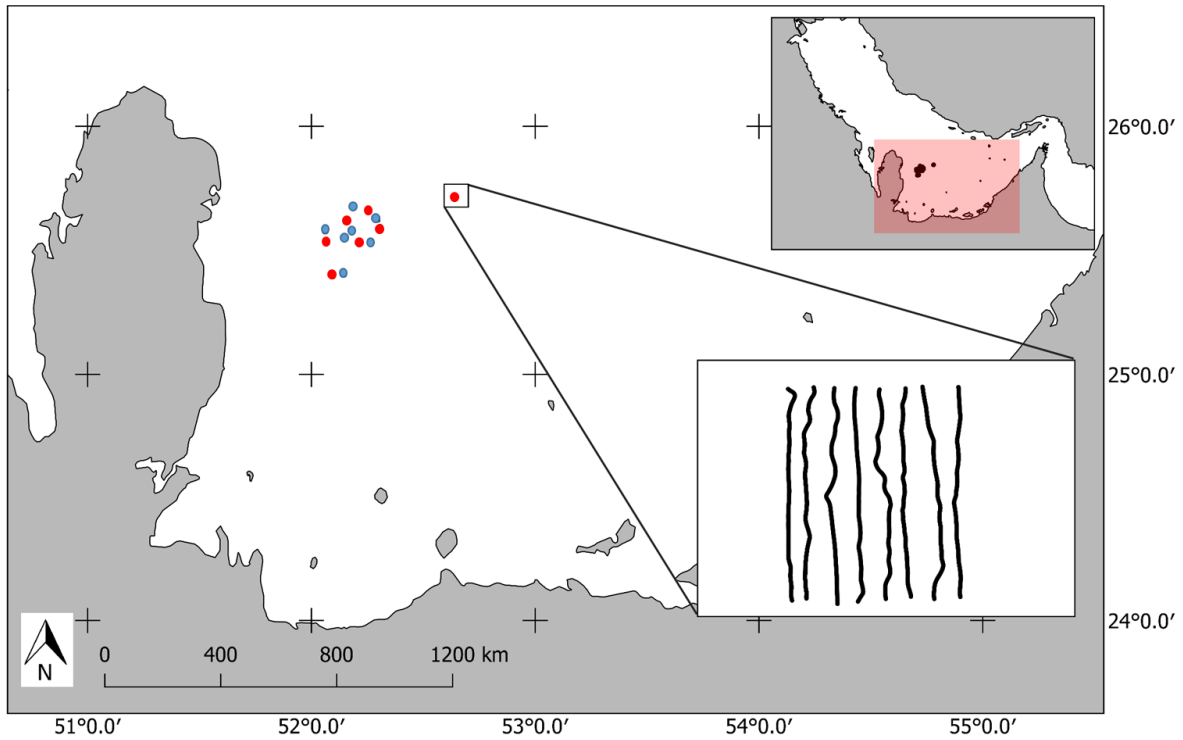
Figure 5.1. Location of the survey sites within Qatari waters. The black dots represent the survey sites and the zoomed in box show the transect lines that are present within each of these, shown with a zoomed in example. Shallow sites shown by red dots and deep sites, blue. The overview map shows the location of Qatar in the Gulf, with the extent of the main map highlighted in red.

Table 5.1 Site locations with groundtruthed habitat type and mean depth (±S.E.M).

Site	Latitude	Longitude	Ground-truthed Habitat	Mean Depth (m) ± S.E.M		Mean VH Habitat value ± S.E.M	
S1	52.0673	25.5312	Reef	13.47±	0.038	1.613±	0.024
S2	52.15353	25.61665	Sand	21.42±	0.023	2.077±	0.043
S3	52.2561	25.65897	Reef	18.95±	0.035	1.934±	0.040
S4	52.21457	25.53402	Reef	17.9±	0.024	1.663±	0.024
S5	52.3058	25.5919	Reef/Sand	18.96±	0.014	2.04±	0.038
S6	52.63953	25.71557	Reef inc Coral	21.89±	0.072	2.091±	0.047
S7	52.09082	25.3949	Reef	16.94±	0.016	1.926±	0.029
D1	52.18418	25.679	Mud	32.17±	0.042	1.685±	0.021
D2	52.29328	25.62682	Mud	32.46±	0.016	1.638±	0.045
D3	52.26392	25.53658	Mud	32.48±	0.007	1.387±	0.019
D4	52.14867	25.55352	Mud	27.58±	0.003	1.596±	0.017
D5	52.06132	25.57995	Mud	28.27±	0.005	1.355±	0.026
D6	52.14158	25.40567	Mud	31.7±	0.007	1.292±	0.012
D7	52.17705	25.58188	Mud/Sand	27.44±	0.028	1.733±	0.025

5.3.2 Equipment

A Biosonics® DTX Split beam echosounder with a 200 kHz transducer was used for the surveys. The transducer was mounted over the port side of the survey vessel (transducer face 1m below surface) as close to the centre of roll and pitch as possible, attached to a pole secured by bespoke brackets. Acoustic data were georeferenced with an integrated Garmin 17Xhvs GPS, and collected with Biosonics acquisition software (Visual Acquisition 6). The circular transducer has a beam opening angle of 6.8°. Pulse duration was 0.4 ms and the specified ping rate was 10 per second. Calibration of the echosounder occurred before the start of the surveys on 03/05/2015 using a Biosonics 36mm Tungsten Carbide 200 kHz Calibration Sphere following the standard methods of Foote et al (1987).

5.3.3 Survey Coverage

In acoustic fish surveys, there needs to be adequate coverage over the survey areas to gain a reliable picture of the fish distribution. Degree of coverage (Λ) is defined as: $\Lambda = D/\sqrt{A}$ where: D is the cruise track length, and; A is the size of the survey area. Empirical data from Aglen (1989) showed the ratio needs to be 6 or over. This was achieved in all the different survey sites with 8 parallel transects covering a survey box of 1km by 1km leading to a Degree of

coverage of 8 in each site. Survey speed was restricted to between 5 and 6 knots (2.5 - 3 m/s) and all surveys were conducted during daylight hours. Surveys occurred from 03/05/17 to 07/05/17.

5.3.4 Data processing

The data were collected with the Biosonics software Visual Acquisition 6 as DT4 files. These files were then converted and post processed with the software package Sonar5 (ver 6.0.4) (Balk and Lindem, 2006). Analysis in Sonar5 (ver 6.0.4) followed the Software Guided Analysis (SGA) routine (based on the Standard Operating Procedure of Parker Stetter et al. 2009) to ensure a consistent approach. The analysis was based up Echo Integration (EI) which divides the sum of reflections from all fish over a segment (the volume backscattering coefficient, Sv) by the mean echo intensity from individual fish (TS) *in situ,* which is derived from the backscattering cross section (σ_{bs}): **TS = 10*Log($\sigma_{bs)}$** (Rudstam et al. 2009; Winfield et al. 2012). Sv (dB) can be converted to the linear form sv (m^2/m^3) through the equation: **sv = 10^(Sv/10)** (Rudstam et al. 2009). Volumetric fish densities (ρ) are therefore calculated as: ρ= **sv/σ_{bs}** The criteria to accept SED were a minimum echo length of 0.8dB a maximum of 1.2dB and a maximum angle standard deviation of 0.8 degrees. Multipeak suppression was set to 'medium' in the software which demands a local dip of 1.5dB between peaks before rejecting the echo. In order to initially separate fish from other particulate targets such as plankton (Parker-Stetter et al. 2009) thresholds of -60dB for SED and -66dB for Sv were applied. Acoustic SED returns below -60dB were therefore excluded by this, and any other remaining noise was removed by eye. A Time Varied Gain (TVG) correction of 40log(R) for TS values and 20log(R) for Sv values are applied by the software as standard (Balk and Lindem 2006). Each 1km transect was not further divided into elementary distance sampling units (EDSUs), to minimise the numbers of cells with no backscattered echo energy (Emmrich et al., 2010). The seabed was automatically detected and manual editing occurred when necessary. In order to ensure that no echoes from the seabed were classified as fish, a bottom margin of 0.5m was applied and data from this layer were not analysed. Similarly, a layer of between 1 and 5m (depending on the sea state) was applied to remove any surface noise. The Nv index (Sawada et al. 1993), was calculated for all transects and all were acceptably low (Nv<0.1) indicating TS estimates were unbiased (Rudstam et al. 2009; Yule et al. 2013).

5.3.5 Fish distribution

In examining fish distribution between sites and habitats the arithmetic mean of transects per each site category of fish density (# individuals per $1000m^3$) and also the area scattering coefficient (Sv) (dB) were investigated. Sv quantifies the sum of fish backscattering cross sections per volume, and is often used as a proxy for biomass (Simmonds and MacLennan, 2005; Boswell et al. 2010). In order to calculate means and for statistical analyses, the linear form 'sv' (m^2/m^3) was used. Statistical analyses were conducted to determine if differences were present in these fish parameters between shallow sites and deep sites, by the use of 2 sample T tests. Data were checked that assumptions of normality and equal variance were satisfied and log transformed if necessary. If these assumptions were still not achieved then nonparametric Mann-Whitney Wilcox tests were used.

5.3.6 Fish size

In order to examine TS from individual fish, fish were tracked in Sonar5 (ver 6.0.4) using the default criteria to define a track of; a minimum of 4 pings, 2 pings gap and gaiting of 0.3m. It is difficult to track individual fish when they occur in dense schools (on occasion the ratio of Sv in tracks to total Sv was <10%), and although in such cases it was possible to gain some fish from the school periphery, the resultant TS's should therefore be thought of as indicative rather than absolute. The multi species equation of Love (1971) was applied to provide estimates of fish length. It provides the following conversion:

TS= (19.1 log10L) – (0.9 log10 f) – 62.0

Where TS = target strength (dB), L = length of the target (cm), and f = the frequency used.

As in this survey the transducer frequency was 200 kHz this equation then becomes:

TS=19.1 log10(L) -64.07

We examined fish size between sites categories both as Mean TS in decibels (dB) and also as length in cm via application of the Love (1971) formula. Differences in fish size class distributions between the 2 depth categories were tested by performing a 2 sample Kolmogorov-Smirnov test.

5.3.7 Fish sampling

A variety of methods were used to sample the fish species present, unfortunately due to logistical constraints it was not possible to conduct the same strategy at each station. SCUBA surveys were conducted at all shallow sites and additionally D5 and D7. SCUBA surveys consisted of a timed search method to quantify the species present and imagery was recorded on GoPro cameras for subsequent analysis. A cut off of 11 minutes was taken as the limit of video analysis, as this was the length of the shortest bottom time, allowing comparable data across the shallow sites where it was collected. Gill nets (2inch mesh) were set at 3 locations (S1, S7 and D5) which consisted of 8 nets of 90m with 2m overlap with a soak time 2.5hrs and set at a depth of 14m. Handlines were utilised to sample fishes at sites S2, S4, S5, S6, S7, D5, and D7. Data from fish traps were also gathered opportunistically at one station (D6).

Tables of the recorded fish species along with the sampling strategy are given in Tables 5.4 and 5.5.

5.3.8 Habitat.

The data were also processed to provide habitat type by the use of the software Visual Habitat (Ver 1) (VH) (Biosonics Inc). Substrate classification in VH uses Principal Components Analysis (PCA) on returning echoes from the seabed and clustering occurs based on similarities of the echo components, resulting in the delineation of areas with similar acoustic properties based on relative hardness and smoothness of the seafloor (Munday et al. 2013). The depth normalisation option was applied in the software using the mean depth across the surveys. Habitat type along 10 ping sections of each transect was placed in one of 3 categories following PCA analysis routine in VH. 3 classes were chosen as the groundtruthing showed 3 main habitats (mud, sand, and reef). The process can be thought of as 'unsupervised' as acoustic data are segmented before being assigned a habitat type identified from groundtruth observations (Calvert et al. 2014). These habitat categories (1, 2 or 3) were then averaged for each transect and site to provide a mean value. During the surveys, the habitat type was confirmed by either the use of towed camera or via SCUBA divers. Data from different acoustic habitat types were plotted against depth and mean habitat values compared with that from the video groundtruthing in order to determine the efficacy of the acoustic method. We also examined how the Biosonics VH software compared to properties of the bottom echo extracted in Sonar 5 (ver 6.0.4). Specifically we extracted the 'attack' and 'decay' of the bottom echo parameters

which correspond to the seabed hardness and roughness respectively (Balk and Lindem, 2006). We subsequently examine how the data from both software packages are capable of explaining the differences in the fish parameters through regression analysis.

5.3.9 Fish association with habitat

Potential association of fish with the different habitat types was investigated by examining the heights of tracked fish in the water column over the different habitats and sites. A spatial join was performed in a GIS (QGIS, ver 1.8) so that mean depth values were provided in a 5m radius buffer around the tracked fish positions. Fish height off seabed was then estimated as seabed depth minus fish depth. We then examined the relationship between fish size and height off seabed between the different site categories through Mann-Whitney Wilcox tests and regression analysis. Due to the same issues of tracking fish in dense schools as mentioned above, this data should however be thought more of an indication of fish depths and sizes rather than absolute values for all fish surveyed.

5.4   Results

5.4.1 Fish Density

Numerical data on fish distribution between sites is given in Appendix 1 (Table 5.3).
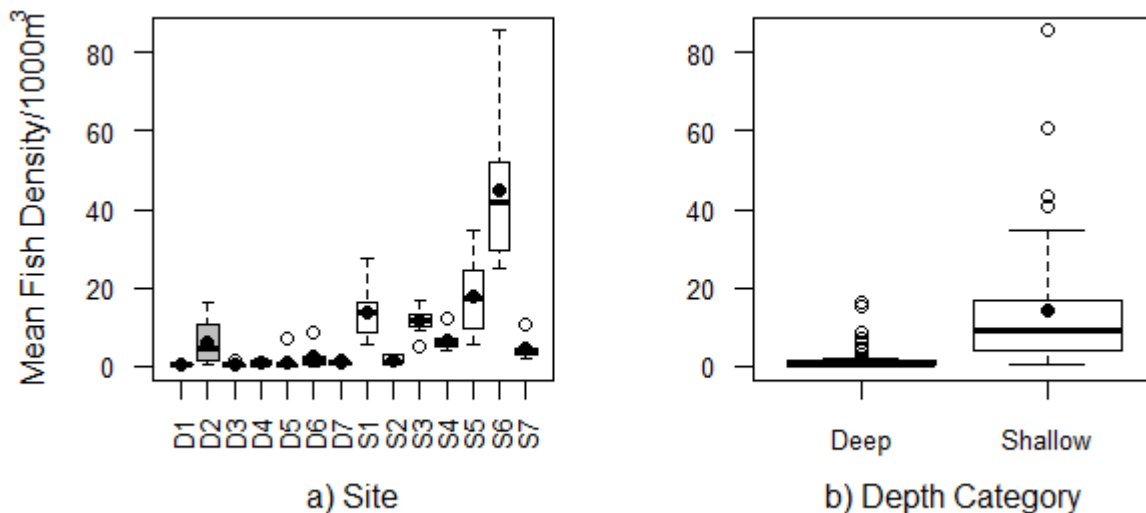
Figure 5.2 Mean fish density expressed as number of fish per 1000m3 at survey transects at each site. Box plots show mean values (black circle), median values (solid horizontal line), and the lower and upper ends of the box are the 25% and 75% quartiles respectively. 'D' are 'Deep' Sites and 'S', shallow. The whiskers indicate 1.5 times the inter-quartile range and points beyond this range are shown by empty circles.

With fish density (in numbers of fish per 1000m^3), a significant difference was detected at the 95% level between transects at shallow category sites and deep category sites by the use of a two-sample t-test (T_{108} = -10.63, P <0.001) with a fourfold greater fish density at the shallow sites (Fig. 5.2).

5.4.2 Biomass (sv)
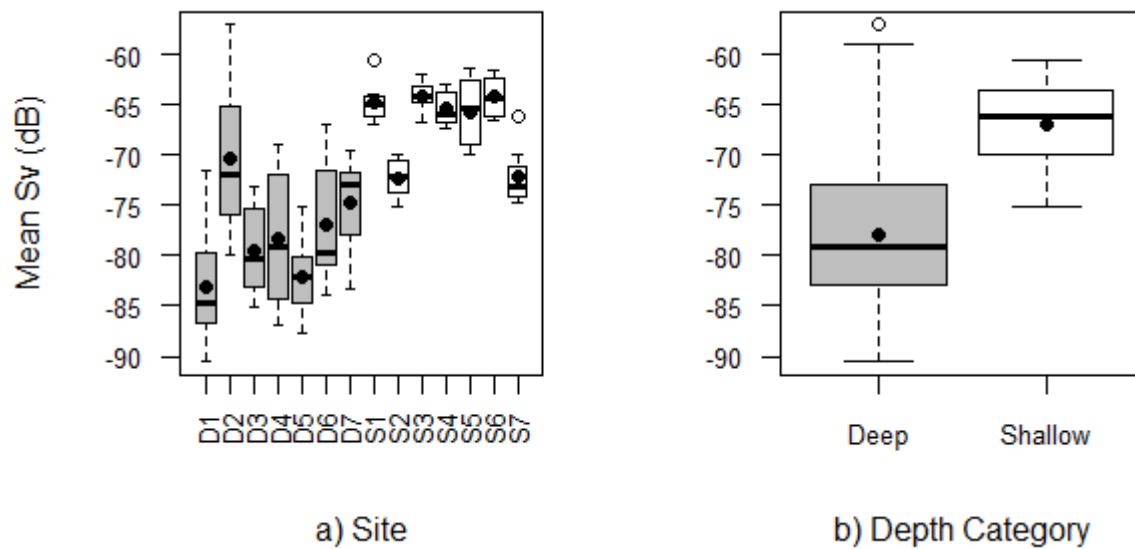
Figure 5.3 The mean scattering coefficient Sv (expressed in dB, for ease of view) a) values per site b) values per depth category. 'D' are 'Deep' Sites and 'S', shallow. See Fig. 5.2 caption for further box plot explanation.

A Mann-Whitney Wilcox test revealed significantly higher values of sv at shallow category transects (W = 293, P <0.001), indicating greater fish biomass values at these sites (Fig. 5.3).

5.4.3 Fish Size
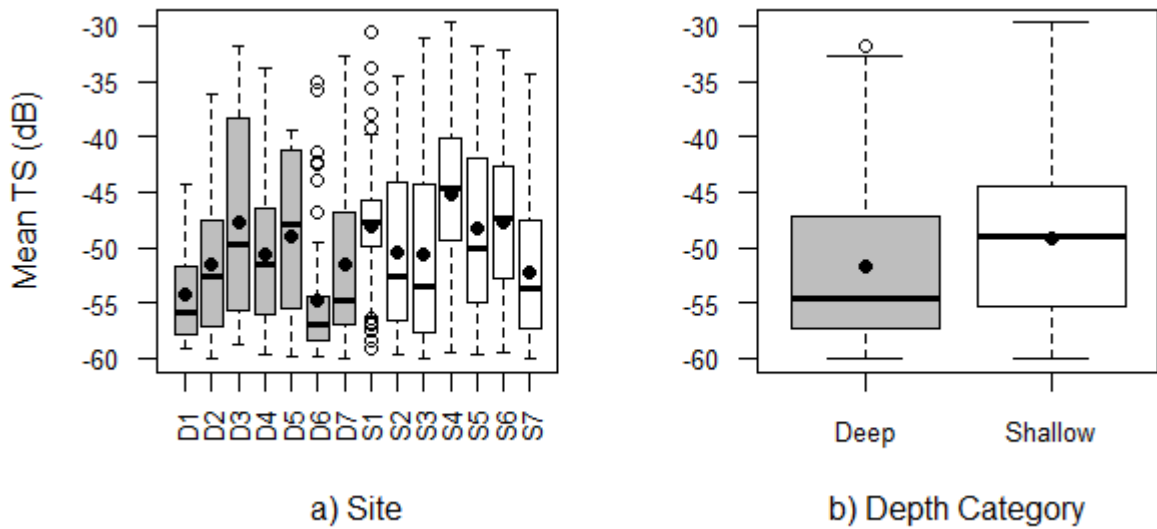
Figure 5.4 The Mean TS values in decibels of fish at a) survey sites b) depth category. 'D' are 'Deep' Sites and 'S', shallow. See Fig. 5.2 caption for further box plot explanation.
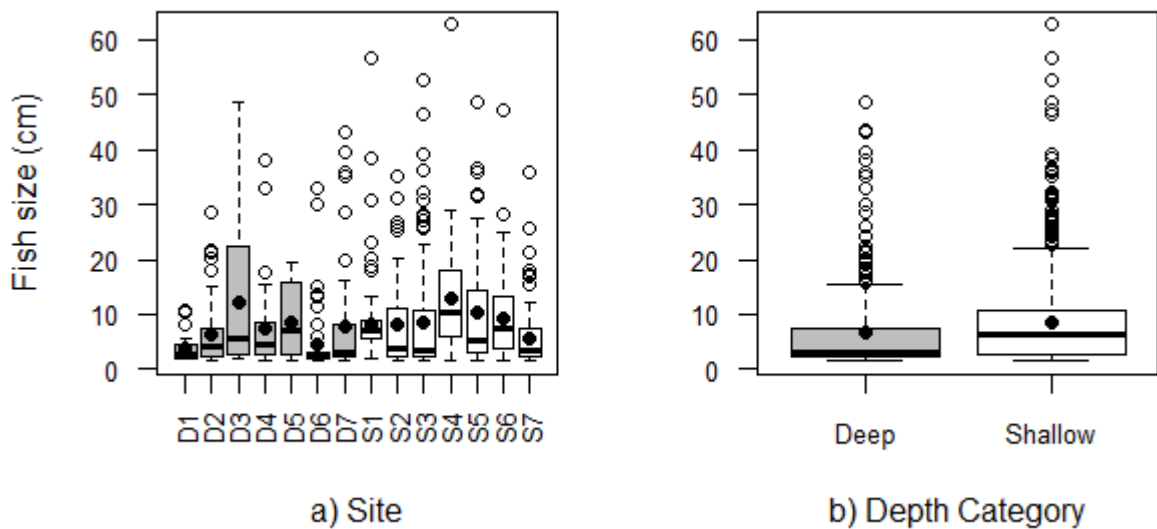
Figure 5.5 The Mean fish size derived using the Love (1971) equation to TS values a) at each survey site, b) at each depth category. 'D' are 'Deep' Sites and 'S', shallow. See Fig. 5.2 caption for further box plot explanation.

Tracked fish had significantly higher values of TS (T_{458}=-5.06, P<0.001) (Fig 5.4); and corresponding fish length (W = 61270, P<0.001) (Fig 5.5) at transects at shallow category sites in comparison to transects at deep category sites. Fish size class distribution data (Figure 5.6)

was also shown to differ significantly between the 2 depth categories using a 2 sample Kolmogorov-Smirnov test (KS test statistic =0.266, KS critical value = 0.097, P<0.05). Data on fish density, biomass and mean fish length are also shown in the bubble plot below (Figure 5.7) to summarise the major findings of this study and to demonstrate how the different parameters are related between sites.
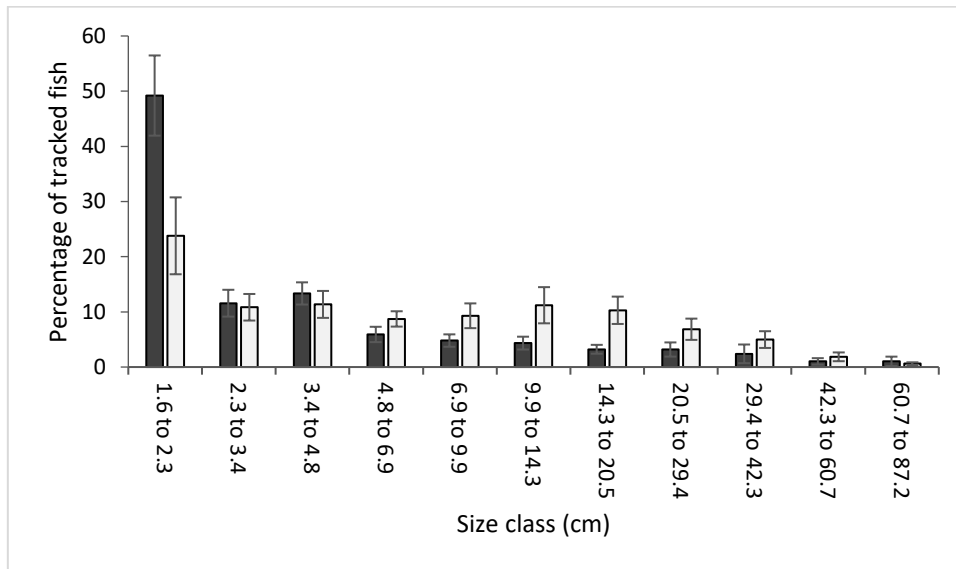
Figure 5.6 Fish size classes gained from applying the Love (1971) equation to TS values of tracked fish along transects at deep (grey) and shallow (white) survey sites. Error bars represent ±SE.
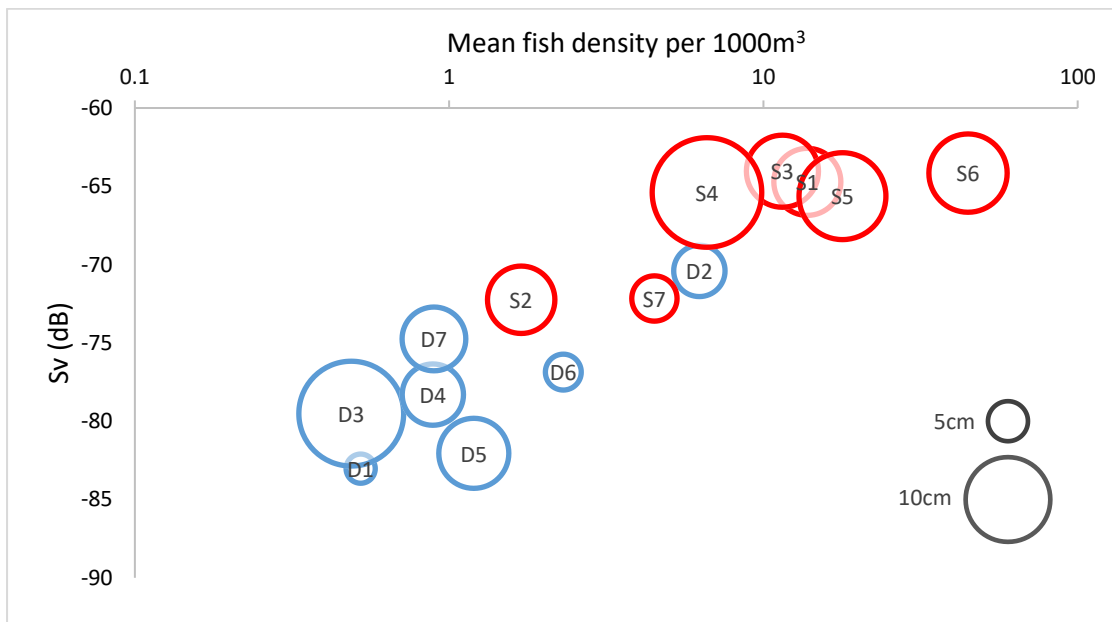
Figure 5.7 Bubble plot summarising the main findings in this study showing the interactions between density (fish/1000m³), size (cm) and the biomass proxy (Sv) (dB). Width of the bubbles represent the mean length of fish from each site. 'D' are 'Deep' Sites and 'S', shallow. Blue circles are from 'Deep' category sites whilst red are from 'Shallow' sites. N.B. Density data are plotted on Log10 scale.

5.4.4 Fish height over seabed
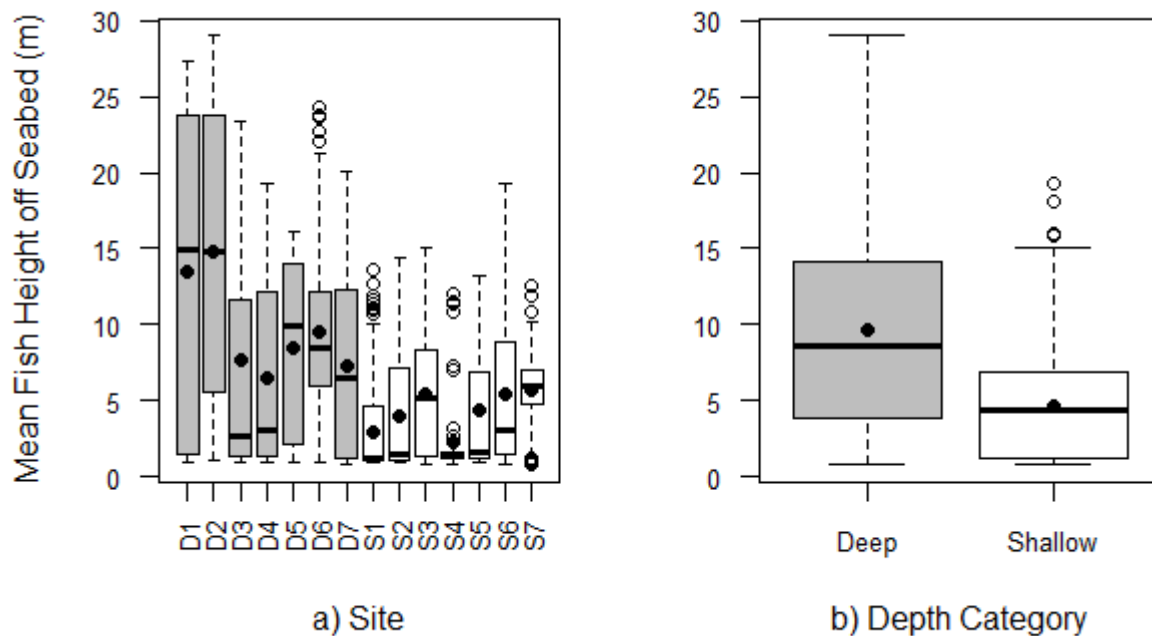
Figure 5.8 Mean values of fish height in the water column a) per site b) per depth category. See Fig. 5.2 caption for box plot explanation.

A Mann-Whitney Wilcox test revealed that the tracked fish were significantly closer to the seabed at shallow sites compared to deep sites (W = 437020, P<0.001), confirming hypothesis (b) (Fig. 5.8). This exploration was taken further by examining fish height above the seabed against fish length for tracked fish at all sites, deep sites and shallow sites (Fig. 5.9). Larger fish (log transformed) were seen to be significantly closer to the seabed across the depth categories; ($F_{1,1569}$= 1010, R^2=0.392, P<0.001) for all sites, ($F_{1,654}$= 139.6, R^2=0.176, P<0.001) for deep category sites and ($F_{1,913}$= 820.9, R^2=0.473, P<0.001) at shallow category sites.
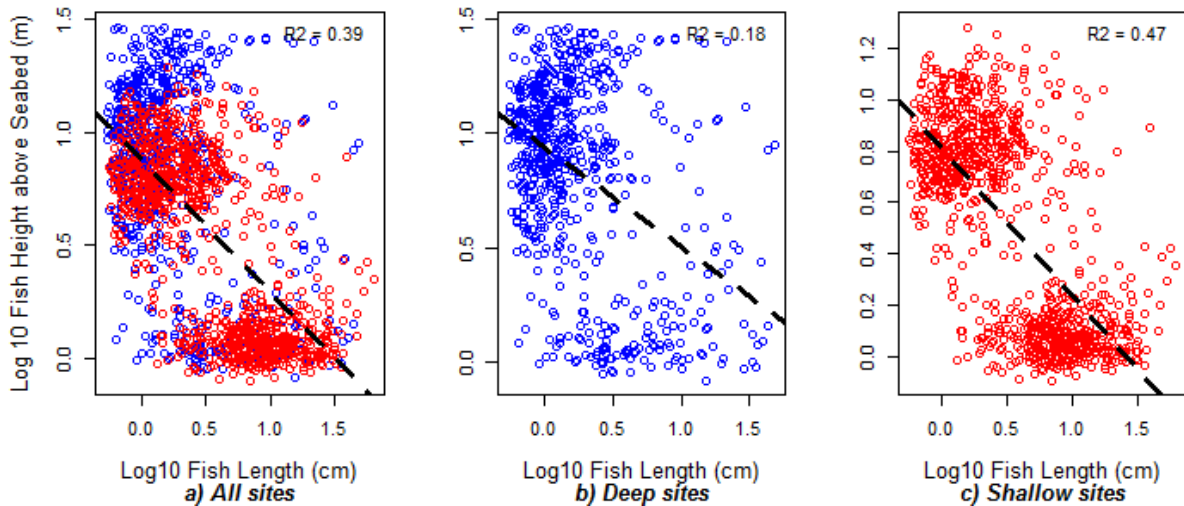
Figure 5.9 Log10 Fish height above seabed (m) plotted against Log10 Fish length (from application of the Love 1971) equation on tracked fish. Blue circles are fish from 'Deep' category sites whilst red are from 'Shallow' sites.

5.4.5 Groundtruthing of fish species

A total of 306 fish were caught during the fishing-based groundtruthing, 230 of these caught at shallow category sites. Across sites the most commonly caught fish species was *Lethrinus borbonicus* which represented 37% of the total catch and of these 95% were caught at shallow sites (35% of total catch). Amongst the deep sites the most commonly caught species was *Carangoides chrysophysis* (6% of total catch, 24% of catch from deep sites), followed by *Diagramma pictum* (5% of total catch, 20% of catch from deep sites).

During the SCUBA surveys 821 individual fishes were recorded. Of these the most commonly recorded fish species was *Lethrinus lentjan* (25% of total individuals recorded) however these were only recorded in high density at one site (S7). The most widely recorded species across sites was *Acanthopagrus bifasciatus* (18% of total individuals recorded). Site S7 had the highest number of individuals recorded (34%) followed by S6 (23%). Full details on the fish species recorded during groundtruthing are given in Appendices 5.8.2 and 5.8.3.

5.4.6 Habitat

In comparison to groundtruth data (Table 5.1), the examination of the VH data revealed that higher mean values were associated with harder and more rugose habitats. All 'deep' category transects were then compared with all 'shallow' category transects and a 2 sample t test was performed which confirmed statistically significant differences between the VH mean habitat

values ($T_{105}=10.48$, P <0.001). Similarly with mean values of Attack ($T_{95}=5.64$, P <0.001), and Decay ($T_{97}=6.68$, P <0.001).
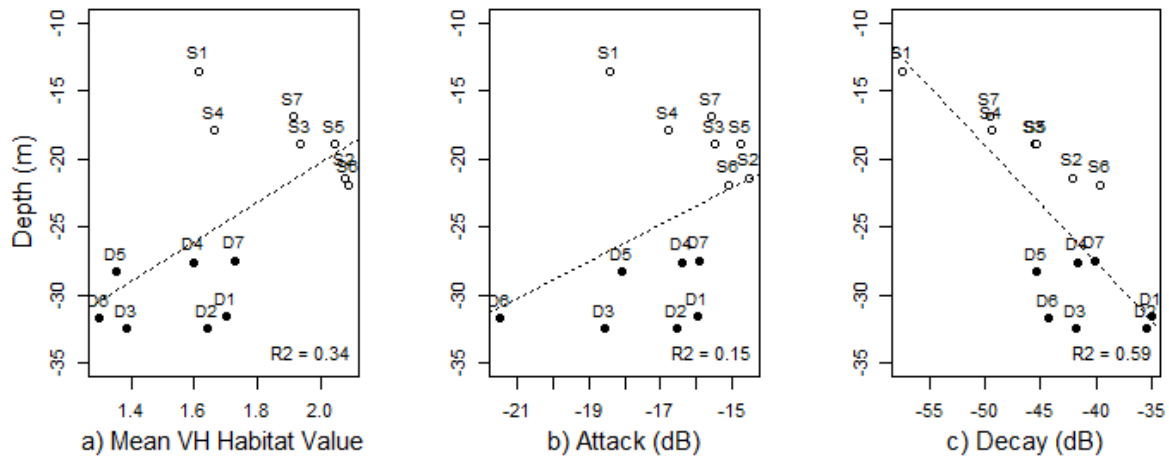
Figure 5.10 Mean values of acoustic data on habitat at the different sites plotted against depth a) Mean Habitat Value from VH, b) mean values of Attack (dB), c) mean values of Decay (dB) 'D' are 'Deep' Sites and 'S', shallow.

Acoustic habitat data were plotted against depth to examine possible correlation (Figure 5.10). There were significant relationships between depth and VH mean habitat ($R^2 =0.34$, $F_{12}=6.24$, P<0.05), and Decay ($R^2 = 0.5916$, $F_{,12}= 17.38$, P<0.05), but not with Attack ($R^2 = 0.1521$, $F_{12}= 2.15$, P= 0.17). It should be noted that with both VH Attack and the opposite trend with depth is displayed when only the shallow sites are examined. Further, Attack (in its linear form) was significantly correlated with VH data ($R^2=0.89$, $F_{12} =93.67$, P <0.05).

5.4.7 Acoustic habitat data for predicting fish distribution

As correlation was seen to occur between the habitat parameters individual regressions were performed rather than multiple regression due to issues of multicollinearity. Mean habitat values (VH, and 'Attack' and 'Decay' from Sonar5 (ver 6.0.4)) per site were plotted against mean site values of fish density and sv, and regression analyses performed (see Table 5.2 for details). This showed that there was a significant relationship with VH class as a predictor of fish density ($R^2= 0.302$, F= 5.20, P< 0.05), but not of biomass (sv) ($R^2= 0.197$, F= 2.94, P= 0.12). The same routine was performed against mean depth values in order to investigate if the acoustic VH habitat results show additional influence over depth alone. There was no significant relationship between depth as a predictor for either fish density or sv.

Table 5.2 Results of regression analysis on acoustic habitat and depth in predicting mean Fish density (number per 1000m³) and sv (m²/m³) (biomass proxy) per site. Regressions that are significant at the 95% level are highlighted in bold.

Variable	Fish/1000m3	sv (m²/m³)
VH Habitat	**R²=0.302, F=5.20, P=0.04**	R²=0.197, F=2.94, P=0.11
Attack	R²=0.098, F=1.31, P=0.28	R²=0.073, F=0.94, P=0.35
Decay	R²=0.004, F=0.05, P=0.84	R²=0.039, F=0.49, P=0.50
Depth	R²=0.165, F=2.37, P=0.15	R²=0.233, F=3.64, P=0.08

5.5 Discussion

5.5.1 Fish distribution between sites

Values of the fish parameters tested (density, sv, TS and corresponding fish length) were all significantly higher at the 'shallow' category oyster bed/reef sites in comparison to the 'deep' category muddier sites. This is in keeping with the behaviour of local fishers who target these areas, mainly by use of fish traps ("gargoor") (Smyth et al. 2016). In other regions, oyster reefs have been also been noted as having higher densities of benthic fishes than sandy habitat (Harding and Mann 1999; Harding and Mann 2001; Lenihan et al. 2001). Habitat complexity plays an important role in structuring ecological communities (Friedlander and Parrish, 1998) and this is likely to have been the case here. The greater structural complexity of the reef habitats at the 'shallow' category sites results in more areas of shelter for fish that are absent from the 'deep' category muddier habitats (Coles and Tarr, 1990), resulting in the higher densities. Generally, the more complex substratum provides habitat for many invertebrates which in turn serve as food resources for many reef fishes (Parrish et al. 1985). This effect of increased habitat rugosity showing greater fish density has been noted by many other authors (Risk, 1972; Luckhurst and Luckhurst, 1978; Öhman and Rajasuriya 1998; Brokovich et al. 2006; Graham and Nash, 2013). Cryptic species, with a close association with the reef matrix will not have been detected by our acoustic methods due to the presence of the 'acoustic dead zone' (Ona and Mitson, 1996) and it is therefore likely that our density estimates are conservative. However, it should be noted that these cryptic species are unlikely to include commercially-targeted species.

Site S6 had the largest mean value of fish density and second highest mean value of sv. This site is also known to have the most complex habitat of the sites with greatest amounts of live

coral in Qatari territorial waters (Rezai et al. 2004; Sheppard et al. 2010) dominated by the genus *Acropora* (Riegl, 1999) and confirmed by diver video. The amount of live coral has also long been known to have a positive relationship on the number of fish species and individuals (Carpenter et al. 1981; Bell and Galzin 1984; Bouchon-Navaro and Bouchon 1989; Graham and Nash, 2013). This area is known to be a highly productive fishing ground (Al-Ansi and Al-Khayat, 1999), which is supported by our results. Of the shallow sites S2 had lowest fish density and here it should be noted that this site was groundtruthed as sand rather than reef, unlike the other shallow and more rugose sites.

In most studies of ecology of reef fishes, depth seems to be an important habitat variable affecting density and distribution (Friedlander and Parrish, 1998) and linear declines in taxonomic diversity have been seen with increased depths (Jankowski et al. 2015). In our study it is most likely that the differing habitat at depth is the main driver in the fish distribution rather than the depth *per se* as demonstrated by higher R^2 values (Table 5.2).

The one site in the 'deep' category sites that stands out as having higher fish density and sv values than the others in this category is site D2. Here, much plankton and schools of (presumably planktivorous) fish were seen on the acoustic record. The patchy nature of the fish schools at this site lead to large variability in the data especially with the sv (biomass proxy) values between transects. The reasons why such a distribution was only observed at this site are unclear and unfortunately were not possible to establish within the scope of the survey.

From the fishing-based groundtruthing the most common fish species caught was *Lethrinus borbonicus*. These are known to be found in sandy areas in proximity to reefs during daytime, and they mainly feed at night over reefs and slopes (Carpenter and Allen, 1989). Other lethrinid species were also regularly seen during the groundtruthing regime and *Lethrinus lentjan* was the most commonly recorded species during the SCUBA diving video surveys. This species is known to inhabit sandy substrates in coastal areas, deep lagoons and near coral reefs (Sommer et al. 1996). It is acknowledged that the differing methods of fish groundtruthing are not quantifiably comparable and it is likely that hand lines and gill nets sample a more pelagic community in comparison to the mainly demersal species seen on the diver video. Due to this and the lack of species-specific TS–Length formulae for many of the species encountered, the multi-species TS–Length formula from Love (1971) was applied, which is likely to have resulted in inaccuracies in fish sizes (Simmonds and MacLennan, 2005). However it does provide a consistent and intuitive relative index from which comparison can be made (Yule,

2000; Boswell et al. 2007). We acknowledge however, that if acoustic returns could have been discerned to a species level (and TS-Length formula were available) then more accurate, length, weight and subsequently biomass estimates (in units such as t/ha) would have been possible. Although fish sampling occurred consecutively with the acoustic surveys, the larger area covered by the acoustics and heterogeneous nature of the community makes quantitative comparison between the datasets problematic. Additionally, and with a less size selective sampling regime, a comparison in acoustic size versus sampled fish size would have been possible.

In examining fish height in the water column, a stronger association with the seabed was shown at the more rugose shallow sites (Thesis Hypothesis 5b). Further, when examined in combination with fish size a clear trend was revealed with the near absence of larger fish higher in the water column, with these being more closely associated with the seabed over both site categories (but with stronger association at shallow category sites). Smaller fish were more ubiquitous throughout the water column. Rugosity has been seen to have an influence on fish size with increased complexity increasing fish size (Friedlander and Parrish, 1998). This is likely due to the larger sized fish mirroring the larger hole sizes in more rugose substrata (Hixon and Beets, 1993). Alternatively this may be due to a greater density of prey for larger fishes, both invertebrates and other fish, over more rugose areas. We acknowledge that diel cycles have a large effect on fish distribution in the water column, with fish tending to be more dispersed during night (e.g. Bohl, 1980). As surveys were all carried out during daylight hours, the data should however be comparable, but night-time surveys may have yielded different results. We are unaware of any other studies examining fish-habitat linkages in this manner and therefore further targeted research would be invaluable.

Variation in fish distribution is also likely to have been introduced by environmental factors that unfortunately were beyond the scope of this study. Other studies have seen effects on fish distribution due to variables such as temperature, salinity and dissolved oxygen (Marshall and Elliott, 1998) and zooplankton (Maravelias et al. 2006) and future studies in the area incorporating these would be valuable. Other sources of unexplained variation could result from ecological or behavioural characteristics of the fish present (Moore et al. 2009). The distribution of fish we encountered could also be related to survey bias in the form of fish avoidance of the survey vessel (De Robertis and Handegard, 2013), which may have had a greater effect at shallower sites (Vabø et al. 2002). This effect may also have manifested itself

differently with different fish sizes, with larger fish exhibiting greater avoidance than smaller fish, although as small fish have previously shown stronger avoidance behaviour (Soria et al. 1996; Draštík and Kubečka, 2005) this is considered unlikely. In freshwater systems using similar size survey vessels to ours, minimal ship avoidance has been reported (Draštík and Kubečka, 2005; Wheeland and Rose, 2015), we therefore expect any ship avoidance effects to be small.

Essential Fish Habitat (EFH) has been defined as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (Rosenburg et al. 2000). This definition however offers no opportunities to distinguish gradations in fish habitat quality (Harding and Mann, 2001). Some authors have previously defined oyster beds as EFH for some species (Breitburg, 1999), whilst others suggest that fish are drawn to oyster beds due to the greater amounts of food present (Harding and Mann, 2001), rather than being 'essential' *per se*. More detailed species-specific habitat use and life history information is required to categorise the shallow sites as EFH. However we have confirmed Thesis Hypothesis 5a, that these shallow oyster bed/reef habitats, harbour significantly higher fish density and larger fish than surrounding areas and are highly important for fish in this region of the Gulf.

5.5.2 Acoustic Determination of Habitat

Developing acoustic monitoring programmes that can integrate habitat attributes and link them to population productivity and biodiversity have been identified as a priority area of research (Anderson et al. 2008). Through processing the acoustic data to additionally give information on habitat, this study has gone some way towards this with time- and cost-saving implications (Freeman et al. 2004; Mackinson et al. 2004; Koslow, 2009). The acoustic habitat data resultant from VH software was seen to be capable of distinguishing between habitat types with shallow reef sites being significantly distinct from the deeper muddy sites. This was also the case with Attack and Decay from Sonar5 (ver 6.0.4). Video data confirmed the 'deep' sites to be comprised of muddy sediments, whereas the 'shallow' sites were generally characterised by hard substrate/reef. Of the 'shallow' category sites S2 was groundtruthed as being more sand rather than reef, but the VH acoustic habitat data didn't separate this site significantly from the other 'shallow' sites, potentially indicating that water depth over the seabed may have had an overriding impact on the habitat clustering (Greenstreet et al. 1997). This may have also been the case with similar VH values between deep and shallow sites when differences in depth were

not great. Further, the shallowest site S1 had acoustically dissimilar habitat from other shallow sites, but the groundtruthing showed this was not the case. It is difficult to determine the relative importance of depth and habitat as across the study area these two parameters are correlated and a thorough study examining similar habitats at different depths and/or different habitats at the same depth would yield valuable information. VH data were processed with depth normalisation applied and a TVG (20logR) was applied in Sonar5 (ver 6.0.4), so acoustic response in theory should not vary with depth. However, other studies have still found a depth-dependency in acoustic habitat data (e.g. Greenstreet et al. 1997; Bax et al. 1999; Foster-Smith et al. 2004; Hutin et al. 2005) and the issue of an increasing acoustic footprint with depth has still not been fully resolved (Hutin et al. 2005). The pattern shown by both VH and Attack from Sonar 5 in the shallow sites with the inverse trend with depth compared to the full dataset is also worthy of future research. Further, the highly significant relationship between VH and Attack may indicate that Sonar 5 may also be of use for habitat mapping. As depth and habitat type were seen to be correlated, it may be possible to create a habitat map of the area by the use of bathymetry alone with depth as a proxy for habitat (Walton et al. 2017). Of the acoustic habitat data, it is worthy to highlight that VH data had a significant relationship with fish density across all sites, which was not the case when using solely depth as a predictor.

5.6   Conclusions

Through hydroacoustic surveys we have seen the importance of the 'oyster beds'/'hairãt' and the coral dominated reef site, for fish and fisheries within the Qatari Gulf, and how the use of inexpensive habitat mapping software for fisheries echosounders may assist with classifying these. In these contexts, hydroacoustics can provide a valuable role in Ecosystem Based Management (EBM) and the approach described in this study could be used to identify candidate MPAs with high densities of large fish in a fast, quantitative and non-destructive manner.

5.7   Acknowledgements

5.8 Appendices

5.8.1 Appendix 1

Table 5.3 Mean values (± Standard Error) of the fish parameters investigated at the different sites in Qatar. 'D' represents deep sites and 'S' for shallow sites, 'sv' is an acoustic proxy for biomass, 'TS': Target Strength and 'Love size' is the fish length in cm from applying the Love (1971) equation.

Site	Fish density (#/1000m³)	±S.E.M	sv (m²/m³)	±S.E.M	Mean TS (dB)	±S.E.M	Love Size (cm)	±S.E.M
D1	0.52	0.10	1.32E-08	8.27E-09	-54.30	1.04	3.86	0.61
D2	6.25	2.17	4.42E-07	2.70E-07	-51.46	0.82	6.34	0.77
D3	0.49	0.14	1.74E-08	5.78E-09	-47.79	1.92	12.32	3.02
D4	0.89	0.30	3.76E-08	1.70E-08	-50.58	1.17	7.45	1.39
D5	1.20	0.82	9.06E-09	3.33E-09	-48.95	1.88	8.43	1.74
D6	2.31	0.98	5.71E-08	3.00E-08	-54.74	0.87	4.61	0.91
D7	0.89	0.12	5.01E-08	1.29E-08	-51.53	1.04	7.73	1.38
S1	13.80	2.44	3.74E-07	7.61E-08	-48.06	0.38	8.10	0.52
S2	1.70	0.42	6.44E-08	9.04E-09	-50.40	1.07	8.14	1.20
S3	11.50	1.24	4.14E-07	4.70E-08	-50.59	0.79	8.66	1.00
S4	6.61	0.90	3.09E-07	4.37E-08	-45.14	0.88	12.85	1.38
S5	17.85	3.52	3.55E-07	8.79E-08	-48.32	0.95	10.32	1.28
S6	44.78	7.08	4.23E-07	6.84E-08	-47.81	0.65	9.36	0.74
S7	4.50	0.93	7.71E-08	2.51E-08	-52.17	0.58	5.66	0.52

5.8.2 Appendix 2a

Table 5.4 Fish species sampled and fishing groundtruthing method.

Site	Name	Count	Fishing method
S1	*Paramoncanthus oblongus*	22	Gill nets
	Stephanolepis diaspros	6	Gill nets
	Lethrinus borbonicus	5	Gill nets
	Lethrinus nebulosus	4	Gill nets
	Chiloscyllium arabicum	4	Gill nets
	Sphyraena toxeuma	2	Gill nets
	Pseudotriacanthus strigilifer	2	Gill nets
	Lutjanus fulviflamma	2	Gill nets
	Sphyraena qenie	1	Gill nets
	Siganus canliculatus	1	Gill nets
	Paradachirus mamoratus	1	Gill nets
	Echeneis naucrates	1	Gill nets
	Bothus pantherinus	1	Gill nets
	Argyrops spinifer	1	Gill nets
S2	*Lethrinus borbonicus*	20	Handline
	Cephalopholis hemistiktos	2	Handline
S4	*Lethrinus borbonicus*	39	Handline
	Lethrinus nebulosus	5	Handline
	Cephalopholis hemistiktos	3	Handline
	Acanthopagrus bifasciatus	4	Handline
	Letherinus lentjan	3	Handline
	Ephephelus bleekeri	1	Handline
S5	*Lethrinus nebulosus*	13	Handline
	Lethrinus borbonicus	8	Handline
	Lenthrinus microdon	4	Handline
	Sphyraena putnamae	1	Handline
	Gnathanodon speciosus	1	Handline
	Epinephelus polylepis	1	Handline
S6	*Lethrinus borbonicus*	34	Handline
	Lethrinus nebulosus	17	Handline
	Cephalopholis hemistiktos	8	Handline
	Epinephelus polylepis	7	Handline
	Abalistes stellatus	1	Handline
S7	*Sphyraena putnamae*	2	Gill nets
	Lethrinus microdon	1	Gill nets
	Siganus canliculatus	1	Gill nets
D5	*Lethrinus borbonicus*	7	Handline
	Lethrinus nebulosus	3	Handline

	Arius thalassinus	2	Handline
	Nemipterus bipunctatus	1	Handline
D6	*Carangoides chrysophysis*	17	Traps
	Diagramma pictum	15	Traps
	Argyrops spinifer	10	Traps
	Lutjanus fulviflamma	2	Traps
	Lethrinus L. nebulosus	2	Traps
	Carangoides bajad	2	Traps
	Saurida tumbil	2	Traps
	Rhabdosargus haffara	1	Traps
	Scomberoides tol	1	Traps
	Thenus orientalis	1	Traps
D7	*Lethrinus nebulosus*	9	Handline
	Carangoides chrysophrys	1	Handline

5.8.3 Appendix 2b

Table 5.5 Fish species recorded on SCUBA Diver video timed search

Site	Species	Count
S1	*Acanthopagrus bifasciatus*	26
	Pomacanthidae spp.	19
	Mullidae spp.	5
	Aethaloperca roga	4
	Lutjanus spp.	3
	Pseudochromis spp.	1
	Amphiprion bicinctus	1
S2	*Acanthopagrus bifasciatus*	13
	Lethrinus nebulosus	5
	Aethaloperca roga	2
S3	*Trachinotus blochii*	44
	Acanthopagrus bifasciatus	21
	Pomacanthidae spp.	13
	Aethaloperca roga	10
	Leptojulis cyanopleura	8
	Pseudorhombus spp.	1
	Amphiprion bicinctus	1
S4	*Lutjanus* spp.	52
	Pomacanthidae spp.	17
	Lethrinus nebulosus	6
	Heniochus acuminatus	2
	Aethaloperca roga	1
	Acanthopagrus bifasciatus	1
S6	*Parupeneus barberinus*	81
	Acanthopagrus bifasciatus	66
	Pomacanthidae spp.	24
	Aethaloperca roga	15
	Leptojulis cyanopleura	6
S7	*Lethrinus lentjan*	203
	Acanthopagrus bifasciatus	24
	Parupeneus barberinus	22
	Pomacanthidae spp.	13
	Mullidae spp.	11
	Aethaloperca roga	4
	Scolopsis ghanam	4
	Heniochus acuminatus	1
D5	*Amblygobius albomaculatus*	2
	Saurida undosquarmis	1
D7	*Lutjanus* spp.	87
	Leptojulis cyanopleura	1

5.9 References

Aglen, A. 1989. Empirical results on precision-effort relationships for acoustic surveys. *ICES Journal of Marine Science*: 1989/B:30, 28 p.

Al-Abdulrazzak D. 2013. Total fishery extractions for Qatar: 1950-2010. pp. 31-37. In: Al-Abdulrazzak D and Pauly D (eds.) From dhows to trawlers: a recent history of fisheries in the Gulf countries, 1950 to 2010. Fisheries Centre Research Reports 21(2). Fisheries Centre, University of British Columbia [ISSN 1198-6727].

Al-Ansi, M., Al-Khayat, J. A. 1999. Preliminary study on coral reef and its associated biota in Qatari waters, Arabian Gulf. *Qatar University Science Journal*, 19 (1999), pp. 294–311

Al-Ansi, M., Priede, I.G. 1996. Expansion of fisheries in Qatar (1980–1992): growth of an artisanal fleet and closure of a trawling company. *Fisheries Research*. *26*, 101-111

Anderson, J.T., Van Holliday, D., Kloser, R., Reid, D.G. and Simard, Y., 2008. Acoustic seabed classification: current practice and future directions. *ICES Journal of Marine Science*, *65*(6), pp.1004-1011.

Balk, H. and Lindem, T. 2006. Sonar 4, Sonar 5, Sonar 6 e Post-processing Systems. Operator Manual. Lindem Data Acquisition, Oslo.

Bax, N., Kloser, R., Williams, A., Gowlett-Holmes, K. and Ryan, T., 1999. Seafloor habitat definition for spatial management in fisheries: a case study on the continental shelf of southeast Australia. *Oceanologica Acta*, *22*(6), pp.705-720.

Bell, J.D. and Galzin, R., 1984. Influence of live coral cover on coral-reef fish communities. *Marine Ecology Progress Series*, 15(3), pp.265-274.

Bohl, E. 1980. Diel pattern of pelagic distribution and feeding in planktivorous fish. *Oecologia 44*: 368-375

Boswell, K.M., Wilson, M.P. and Wilson, C.A., 2007. Hydroacoustics as a tool for assessing fish biomass and size distribution associated with discrete shallow water estuarine habitats in Louisiana. *Estuaries and Coasts*, *30*(4), pp.607-617.

Boswell, K.M., Wells, R.J., Cowan Jr, J.H. and Wilson, C.A., 2010. Biomass, density, and size distributions of fishes associated with a large-scale artificial reef complex in the Gulf of Mexico. *Bulletin of Marine Science*, *86*(4), pp.879-889.

Bouchon-Navaro, Y. and Bouchon, C., 1989. Correlations between chaetodontid fishes and coral communities of the Gulf of Aqaba (Red Sea). *Environmental Biology of Fishes*, *25*(1), pp.47-60.

Breitburg, D. L. 1999. Are three-dimensional structure and healthy oyster populations the keys to an ecologically interesting and important fish community? In Luckenbach, M. W., R. Mann, and J. A. Wesson, editors. Oyster reef habitat restoration: a synopsis and synthesis of approaches. Virginia Institute of Marine Science Press. Gloucester Point, Virginia. pp. 239–250

Brokovich, E., Baranes, A. and Goren, M., 2006. Habitat structure determines coral reef fish assemblages at the northern tip of the Red Sea. *Ecological Indicators*, *6*(3), pp.494-507.

Brown, C.J., Smith, S.J., Lawton, P. and Anderson, J.T., 2011. Benthic habitat mapping: a review of progress towards improved understanding of the spatial ecology of the seafloor using acoustic techniques. *Estuarine, Coastal and Shelf Science*, *92*(3), pp.502-520.

Carpenter, K.E. and G.R. Allen. 1989. FAO Species Catalogue. Vol. 9. Emperor fishes and large-eye breams of the world (family Lethrinidae). An annotated and illustrated catalogue of lethrinid species known to date. FAO Fish. Synop. 125(9):118 p. Rome: FAO.

Carpenter, K.E. et al. 1981. The influence of substrate structure on the local abundance and diversity of Philippine reef fishes. In: Proceedings of the 4th International Coral Reef Symposium (Vol. 2, pp. 497-502). Penang, Malaysia.

Calvert, J., Strong, J.A., Service, M., McGonigle, C. and Quinn, R., 2014. An evaluation of supervised and unsupervised classification techniques for marine benthic habitat mapping using multibeam echosounder data. *ICES Journal of Marine Science*, *72*(5), pp.1498-1513.

Coles, S.L. and Tarr, B.A., 1990. Reef fish assemblages in the western Arabian Gulf: a geographically isolated population in an extreme environment. *Bulletin of Marine Science*, *47*(3), pp.696-720.

De Robertis, A. and Handegard, N.O., 2012. Fish avoidance of research vessels and the efficacy of noise-reduced vessels: a review. *ICES Journal of Marine Science*, *70*(1), pp.34-45.

De Young, C. (ed.) 2006. Review of the state of world marine capture fisheries management: Indian Ocean. FAO Fisheries Technical Paper. No. 488. Rome, FAO. 2006. 458p.

Diaz, R.J., Solan, M. and Valente, R.M., 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management*, *73*(3), pp.165-181.

Draštík, V. and Kubečka, J., 2005. Fish avoidance of acoustic survey boat in shallow waters. *Fisheries Research*, *72*(2), pp.219-228.

Ellis, J.R., Rogers, S.I. and Freeman, S.M., 2000. Demersal assemblages in the Irish Sea, St George's channel and Bristol channel. *Estuarine, Coastal and Shelf Science*, *51*(3), pp.299-315.

Emmrich, M., Helland, I.P., Busch, S., Schiller, S. and Mehner, T., 2010. Hydroacoustic estimates of fish densities in comparison with stratified pelagic trawl sampling in two deep, coregonid-dominated lakes. *Fisheries Research*, *105*(3), pp.178-186.

Feary, D.A., Burt, J.A. and Bartholomew, A., 2011. Artificial marine habitats in the Arabian Gulf: review of current use, benefits and management implications. *Ocean and Coastal Management*, *54*(10), pp.742-749.

Feidi, I.H. 1998. Fisheries development in the Arab World. Transformations of Middle Eastern natural environments: legacies and lessons (ed. by J. Albert, M. Bernhardsson and R.

Kenna) pp. 388–406. Yale School of Forestry and Environmental Studies: Bulletin Number 103. Yale University, New Haven, CT.

Foote, K. G., Knudsen, H. P., Vestnes, G., MacLennan, D. N., and Simmonds. E. J. 1987. Calibration of acoustic instruments for fish density estimation: a practical guide. Cooperative Research Report. *Const. Int. Explor. Mer, 144*, 69 pp

Foster-Smith, R.L. and Sotheran, I.S., 2003. Mapping marine benthic biotopes using acoustic ground discrimination systems. *International Journal of Remote Sensing*, *24*(13), pp.2761-2784.

Foster-Smith, R.L., Brown, C.J., Meadows, W.J., White, W.H. and Limpenny, D.S., 2004. Mapping seabed biotopes at two spatial scales in the eastern English Channel. Part 2. Comparison of two acoustic ground discrimination systems. *Journal of the Marine Biological Association of the United Kingdom*, *84*(3), pp.489-500.

Freeman, S., Mackinson, S. and Flatt, R., 2004. Diel patterns in the habitat utilisation of sandeels revealed using integrated acoustic surveys. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.141-154.

Friedlander, A.M. and Parrish, J.D., 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *Journal of Experimental Marine Biology and Ecology*, *224*(1), pp.1-30.

Freitas, R., Rodrigues, A.M., Morris, E., Perez-Llorens, J.L. and Quintino, V., 2008. Single-beam acoustic ground discrimination of shallow water habitats: 50kHz or 200kHz frequency survey?. *Estuarine, Coastal and Shelf Science*, 78(4), pp.613-622.

Graham, N.A.J., Nash, K.I. 2013. The importance of structural complexity in coral reef ecosystems. *Coral Reefs 32*: 315−326

Greenstreet, S.P., Tuck, I.D., Grewar, G.N., Armstrong, E., Reid, D.G. and Wright, P.J., 1997. An assessment of the acoustic survey technique, RoxAnn, as a means of mapping seabed habitat. *ICES Journal of Marine Science*, *54*(5), pp.939-959.

Harding, J.M. and Mann, R. 2001. Oyster reefs as fish habitat: Opportunistic use of restored reefs by transient fishes. *Journal of Shellfish Research*, *20*(3), pp.951-959.

Harding, J.M. and Mann, R., 1999. Fish species richness in relation to restored oyster reefs, Piankatank River, Virginia. *Bulletin of Marine Science*, *65*(1), pp.289-299.

Hixon, M.A. and Beets, J.P., 1993. Predation, prey refuges, and the structure of coral-reef fish assemblages. *Ecological Monographs*, *63*(1), pp.77-101.

Hutin, E., Simard, Y. and Archambault, P., 2005. Acoustic detection of a scallop bed from a single-beam echosounder in the St. Lawrence. *ICES Journal of Marine Science*, *62*(5), pp.966-983.

Hrabik, T.R., Jensen, O.P., Martell, S.J., Walters, C.J. and Kitchell, J.F., 2006. Diel vertical migration in the Lake Superior pelagic community. I. Changes in vertical migration of

coregonids in response to varying predation risk. *Canadian Journal of Fisheries and Aquatic Sciences*, *63*(10), pp.2286-2295.

Jankowski, M.W., Graham, N.A.J. and Jones, G.P., 2015. Depth gradients in diversity, distribution and habitat specialisation in coral reef fishes: implications for the depth-refuge hypothesis. *Marine Ecology Progress Series*, *540*, pp.203-215.

Jensen, O.P., Hansson, S., Didrikas, T., Stockwell, J.D., Hrabik, T.R., Axenrot, T. and Kitchell, J.F., 2011. Foraging, bioenergetic and predation constraints on diel vertical migration: field observations and modelling of reverse migration by young-of-the-year herring Clupea harengus. *Journal of Fish Biology*, *78*(2), pp.449-465.

Khan, N.Y. 2007. Multiple stressors and ecosystem-based management in the Gulf', *Aquatic Ecosystem Health and Management*, *10*: 3, 259-267.

Koslow, J. A. 2009. The role of acoustics in ecosystem-based fishery management. *ICES J Mar Sci*, 66: 966–973.

Kostylev, V.E., Todd, B.J., Fader, G.B., Courtney, R.C., Cameron, G.D. and Pickrill, R.A., 2001. Benthic habitat mapping on the Scotian Shelf based on multibeam bathymetry, surficial geology and sea floor photographs. *Marine Ecology Progress Series*, *219*, pp.121-137.

Larkin, P.A., 1996. Concepts and issues in marine ecosystem management. *Reviews in fish Biology and Fisheries*, *6*(2), pp.139-164.

Lenihan, H.S., Peterson, C.H., Byers, J.E., Grabowski, J.H., Thayer, G.W. and Colby, D.R., 2001. Cascading of habitat degradation: oyster reefs invaded by refugee fishes escaping stress. *Ecological Applications*, *11*(3), pp.764-782.

Love, R.H., 1971. Measurements of fish target strength: a review. *Fisheries Bulletin*, *69*(4), pp.703-715.

Luckhurst, B.E. and Luckhurst, K., 1978. Analysis of the influence of substrate variables on coral reef fish communities. *Marine Biology*, *49*(4), pp.317-323.

Mackinson, S., Freeman, S., Flatt, R. and Meadows, B., 2004. Improved acoustic surveys that save time and money: integrating fisheries and ground-discrimination acoustic technologies. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.129-140.

Maravelias, C.D., Reid, D.G. and Swartzman, G., 2000. Seabed substrate, water depth and zooplankton as determinants of the prespawning spatial aggregation of North Atlantic herring. *Marine Ecology Progress Series*, pp.249-259.

Marshall, S. and Elliott, M., 1998. Environmental influences on the fish assemblage of the Humber estuary, UK. *Estuarine, Coastal and Shelf Science*, *46*(2), pp.175-184.

McConnaughey, R.A., Syrjala, S.E. 2009. Statistical relationships between the distributions of groundfish and crabs in the eastern Bering Sea and processed returns from a single-beam echosounder. *ICES Journal of Marine Science*, *66*: 1425 –1432

Minns, C.K., Moore J.E. 2003. Assessment of net change of productive capacity of fish habitats: the role of uncertainty and complexity in decision making. *Canadian Journal of Fisheries and Aquatic Sciences,* 60:100-116

Moore, C.H., Harvey, E.S. and Van Niel, K.P., 2009. Spatial prediction of demersal fish distributions: enhancing our understanding of species–environment relationships. *ICES Journal of Marine Science*, 66(9), pp.2068-2075.

Munday, E., Moore, B., Burczynski, J. 2013. "Hydroacoustic Mapping System for Quantitative Identification of Aquatic Macrophytes, Substrate Composition, and Shallow Water Bathymetric Surveying." Paper presented at the OCEANS IEEE Conference, San Diego, CA

Öhman, M.C., Rajasuriya, A. 1998. Relationships between habitat structure and fish communities on coral. *Environmental Biology of Fishes*. Volume: *53* (1), pp. 19-31.

Ona, E. and Mitson, R.B., 1996. Acoustic sampling and signal processing near the seabed: the deadzone revisited. *ICES Journal of Marine Science*, *53*(4), pp.677-690.

Parker-Stetter, S.L. Rudstam, L.G., Sullivan, P.J., Warner, D.M. 2009. Standard operating procedures for fisheries acoustic surveys in the Great Lakes. Great Lakes Fisheries Commission Special Publication 09-01

Parrish, J.D., Callahan, M.W. and Norris, J.E., 1985, May. Fish trophic relationships that structure reef communities. In *Proceedings of the 5th International Coral Reef Congress* (Vol. 4, pp. 73-78).

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R. and Zeller, D., 2002. Towards sustainability in world fisheries. *Nature* 418 (6898): 689-695.

QGIS Development Team, 2016. QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://www.qgis.org/

Rezai, H. et al. 2004. Coral Reef Status in the Ropme Sea Area: Arabian/Persian Gulf, Gulf Of Oman and Arabian Sea. p: 155-170 in C. Wilkinson (ed.). Status of coral reefs of the world: 2004. Volume 1. Australian Institute of Marine Science, Townsville, Queensland, Australia. 301 p

Riegl, B. 1999. Corals in a non-reef setting in the southern Arabian Gulf (Dubai, UAE): fauna and community structure in response to recurring mass mortality. *Coral Reefs*, Volume: *18*(1), 63-73.

Risk, M.J. 1972. Fish diversity on a Coral Reef in the Virgin Islands. *Atoll Research Bulletin*, Volume *193*, pp. 1-6.

Rose, K.A., 2000. Why are quantitative relationships between environmental quality and fish populations so elusive?. *Ecological Applications*, *10*(2), pp.367-385.

Rosenberg, A., Bigford, T.E., Leathery, S., Hill, R.L. and Bickers, K., 2000. Ecosystem approaches to fishery management through essential fish habitat. *Bulletin of Marine Science*, *66*(3), pp.535-542.

Rudstam, L.G., Parker-Stetter, S.L., Sullivan, P.J. and Warner, D.M., 2009. Towards a standard operating procedure for fishery acoustic surveys in the Laurentian Great Lakes, North America. *ICES Journal of Marine Science*, *66*(6), pp.1391-1397.

Sale, P.F., Feary, D.A., Burt, J.A., Bauman, A.G., Cavalcante, G.H., Drouillard, K.G., Kjerfve, B., Marquis, E., Trick, C.G., Usseglio, P. and Lavieren, H.V., 2011. The growing need for sustainable ecological management of marine communities of the Persian Gulf. *AMBIO: A Journal of the Human Environment*, *40*(1), pp.4-17.

Sawada, K., Furusawa. M., Williamson. N.J., 1993. Conditions for the precise measurement of fish target strength *in situ*. *Journal of the Marine Acoustical Society*. *20*; 15–21.

Sheppard, C., Al-Husiani, M., Al-Jamali, F., Al-Yamani, F., Baldwin, R., Bishop, J., Benzoni, F., Dutrieux, E., Dulvy, N.K., Durvasula, S.R.V. and Jones, D.A., 2010. The Gulf: a young sea in decline. *Marine Pollution Bulletin*, 60(1), pp.13-38.

Siddeek, M.S.M., Fouda, M.M. and Hermosa, G.V., 1999. Demersal fisheries of the Arabian Sea, the Gulf of Oman and the Arabian Gulf. *Estuarine, Coastal and Shelf Science*, *49*, pp.87-97.

Simmonds, J. and MacLennan, D.N., 2005. *Fisheries acoustics: theory and practice*. John Wiley and Sons.

Sivasubramaniam, K., Ibrahim A.M. 1982. Demersal Fish Resources Around Qatar. *Qatar University Science Bulletin*. 2: 305-352.

Smyth, D., Al-Maslamani, I., Chatting, M. and Giraldes, B., 2016. Benthic surveys of the historic pearl oyster beds of Qatar reveal a dramatic ecological change. *Marine Pollution Bulletin*, *113*(1), pp.147-155.

Sommer, 2003. State of the marine environment report. ROPME/GC-11/003. Regional Organization for the Protection of the Marine Environment, Kuwait.

Sommer, C. et al. 1996. FAO species identification field guide for fishery purposes. The living marine resources of Somalia. FAO, Rome. 376 p

Soria, M., Fréon, P. and Gerlotto, F., 1996. Analysis of vessel influence on spatial behaviour of fish schools using a multi-beam sonar and consequences for biomass estimates by echo-sounder. *ICES Journal of Marine Science*, *53*(2), pp.453-458.

Stamatopoulos, C. and Abdallah, M., 2015. Standardization of Fishing Effort in Qatar Fisheries: Methodology and Case Studies. *Journal of Marine Science. Research and Development*, *5*(3), p.1.

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C., 2011. Underwater acoustics for ecosystem-based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, *442*, pp.285-301.

van der Kooij, J., Kupschus, S. and Scott, B.E., 2011. Delineating the habitat of demersal fish assemblages with acoustic seabed technologies. *ICES Journal of Marine Science*, *68*(9), pp.1973-1985.

Vabø, R., Olsen, K. and Huse, I., 2002. The effect of vessel avoidance of wintering Norwegian spring spawning herring. *Fisheries Research*, *58*(1), pp.59-77.

Wheeland, L.J. and Rose, G.A., 2015. Acoustic measures of lake community size spectra. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.557-564.

Walton, M.E., Hayes, J., Al-Ansi, M., Abdallah, M., Al Maslamani, I., Al-Mohannadi, M., Al-Shaikh, I., D'Urban Jackson, T., Szostek, C., Egerton, J. and Kaiser, M.J., 2017. Towards spatial management of fisheries in the Gulf: benthic diversity, habitat and fish distributions from Qatari waters. *ICES Journal of Marine Science*. In press

Winfield, I.J., Emmrich, M., Guillard, J., Mehner, T. and Rustadbakken, A., 2011. WISER Deliverable 3.4-3: Guidelines for standardisation of hydroacoustic methods**. *WISER Project Newsletter*, *2*, pp.4-4.

Yule, D.L., 2000. Comparison of horizontal acoustic and purse-seine estimates of salmonid densities and sizes in eleven Wyoming waters. *North American Journal of Fisheries Management*, *20*(3), pp.759-775.

Yule, D.L., Adams, J.V., Hrabik, T.R., Vinson, M.R., Woiak, Z. and Ahrenstorff, T.D., 2013. Use of classification trees to apportion single echo detections to species: application to the pelagic fish community of Lake Superior. *Fisheries research*, *140*, pp.123-132.

6 Chapter 6. Hydroacoustics for examining fish size spectra in marine environments

6.1 Abstract

The use hydroacoustics is investigated to derive Normalised Biomass Size Spectra (NBSS) at three case studies locations: The Cayman Islands, Mexico and Qatar. This approach examines the size structure of the fish population to derive metrics from the relative abundance of larger vs smaller fish. As hydroacoustic methods are able to sample fish throughout the fish size spectrum, which is a challenge for traditional methods, they have the potential to give a more complete description of the NBSS. Through this approach we find that more complex habitats had significantly steeper NBSS slopes and higher midpoint heights at sites in the Qatari waters of the Arabian Gulf. Higher midpoints are also seen in more complex habitats afforded protection from fishing pressure in Mexico. Most sites revealed that NBSS had significantly better fits with quadratic models as opposed to linear models, with best quadratic fits occurring with most complex habitat in combination with protection. It appears that an unfished community may therefore demonstrate that NBSS has a domed/quadratic structure, whereas a more exploited community, with a higher abundance of small fish may show more linear patterns. Sites with higher structural complexity also had greater NBSS curvature between fished sites. We also examine differences in resultant size spectra from different acoustic processing methods, which generally showed good agreement. A hydroacoustic approach to NBSS, which is novel in the marine environment, helps to provide further insight into the nature of size spectra and additionally may represent a valuable tool for the monitoring of marine fish communities.

6.2 Introduction

The quantification of fish size spectra is a field that shows much promise; it can be used as an indicator of fisheries exploitation and may give fruitful insight into fisheries management and conservation measures (Zwanenburg, 2000; Jiang et al. 2009, Pollom and Rose, 2015). In the marine ecosystem, size is the single most important parameter governing biological processes (Andersen et al. 2015) and this is often a better indicator of trophic level than species identity (Jennings 2005). There is an inverse relationship between the abundance of organisms and their size because predators typically feed on prey smaller than themselves (Sprules and Barth, 2015). There may also be differences due to size-based mortality or differing sizes of individuals between habitats (e.g. smaller fish in nursery areas). Plotting the relation between abundances (or biomass) of fish per each size or weight class (logged) allows a representation of the relative abundances of different sized fish (Trebilico et al. 2013). The resultant intercepts and midpoint heights reflect the level of productivity and abundance respectively (Bianchi et al. 2000; Daan et al. 2005; Sweeting et al. 2009). Whilst the slopes depend on the level of exploitation due to the relative abundances of small vs large fish (Gislason and Rice 1998; Zwanenburg, 2000; Yemane et al. 2010). This is because fishing pressure tends to reduce the abundance of larger-bodied individuals more than that of smaller individuals (Jennings and Kaiser 1998), leading to steeper slopes being associated with higher levels of fishing pressure (Pope and Knights, 1982; Blanchard et al. 2009; Robinson et al. 2016) (Figure 6.10). Intercepts can also be influenced by exploitation level (Yemane et al. 2010) as the two parameters are correlated (Sweeting et al. 2009) and for this reason we focus on midpoint heights. The size-based approaches that determine the impacts of fishing intensity on fish assemblages and ecosystem health (Rice, 2000) are becoming increasingly important as scientists are progressively switching their attention from single species-based approaches to more ecosystem-based approaches for management and assessment (Bianci et al. 2000; Trenkel et al. 2011). Size spectra analyses provide a general and powerful way to assess ecosystem-level impacts of human- and/or environment-driven changes (Wilson et al. 2010; Blanchard et al. 2017).
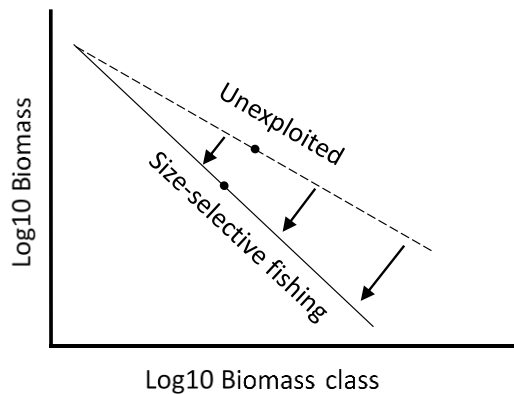
Figure 6.1. *How the slope of size spectra from unexploited community (dashed line), alters as a result of size-selective fishing (solid line). The black circles show how the midpoint height changes from unexploited size spectra with size selective fishing. Adapted from Sweeting et al. (2009).*

Due to the effects of fishing on the nature of Normalised Biomass Size Spectra (NBSS), the differences between NBSS in Marine Protected Areas (MPAs) and control areas has been the subject of much research. McClanahan and Graham (2005) found only a weak positive rise in the slope in the youngest marine park over time (out of 4 examined) (due to high variability between UVC transects) but state that further use of size spectra analysis across gradients of fishing pressure will improve understanding of the size-based implications of fishing and difference in management regimes on coral reefs. Friedlander et al. (2007), found significantly higher intercept values and less steep slope (but not significantly so) at protected sites in Hawaii. The greater size-spectra height due to reduced fishing pressure on abundance has also been seen in MPAs in the Seychelles (Graham et al. 2007). Responses to MPA creation may not actually be an increase in fish abundance but larger size classes of fishes and a greater fish biomass (Edgar and Stuart-Smith, 2009), which would be reflected in the slope steepness. Size spectra slopes are also sensitive to the effects of habitat becoming steeper with increasing habitat complexity, largely as a result of a greater abundance of small fish (Graham et al. 2007; Wilson et al. 2010; Alvarez-Filip et al. 2011; Rogers et al. 2014). Midpoint heights and intercepts have been seen to increase with increasing complexity (Wilson et al. 2010; Alvarez-Filip et al. 2011, respectively). Our case studies allow us to examine the effects of both fishing protection and differing habitat type on hydroacoustically-derived NBSS indices. It should be noted that the case studies described herein were not designed specifically for this purpose, but

it is hoped that this study may prompt future studies to further test such technologies and techniques in the marine environment.

Size spectra analyses are only applicable over the size range at which the gear effectively samples the community (Jennings and Dulvy, 2005; Yurista et al. 2014). In surveys of fish size spectra this bias manifests itself most commonly through the mesh size used in the fishing nets (Shin et al. 2005). The other common method to derive NBSS is through Underwater Visual Census (UVC). However, size spectra derived from different UVC methods might vary substantially (Robinson et al. 2016), and UVC methods can introduce error in the counts of small or large fish size classes (McClanahan and Graham, 2005; Bozec et al. 2011, Ward-Paige et al. 2010). Therefore deriving size spectra data using acoustic techniques removes many of these biases. Further, size-based approaches which use automated survey instruments can reduce the survey effort, allowing large sample sizes and greater spatial and temporal resolution in comparison to traditional methods (Sprules and Barth, 2015). In this manner hydroacoustic methods can be used to produce size-based information that can be utilised in abundance spectra ecosystem models. To date, however, most work has focused on closed freshwater systems (Brandt et al. 1991; Yule et al. 2013; Yurista et al. 2014, Pollom and Rose, 2015, de Kerckhove et al. 2015; Wheeland and Rose, 2015). The application of acoustic size spectra in the marine environment is however not represented in the literature, despite its recognised potential (Trenkel et al. 2011).

Most authors state that normalised size spectra best described by linear models (Boudreau and Dickie 1992; Jennings and Mackinson, 2003) however others suggest the size spectrum is better described by a quadratic model over a linear one (Sprules and Barth, 2015). Some suggest that the degree of curvature of a quadratic model is more affected by fishing pressure, with increased fishing increasing the curvature, rather than the linear model slope (Rochet et al. 1999, Benoit and Rochet, 2004, Shin and Cury, 2004). Further it has been stated that a simple linear model may not be a complete description of the size distribution of a community (Sprules and Barth, 2015). We test the hypothesis that our hydroacoustic data will fit quadratic models better than linear models at our case study sites (Thesis Hypothesis 7).

Whilst size spectra analyses are routinely conducted across ecosystems to determine relative abundances of different trophic groups, we also examined biomass spectra at single species (Nassau Grouper – *Epinephelus striatus*) aggregations in the Cayman Islands. Although this

exercise does not give any information about trophic levels, this was conducted to see if NBSS could reveal similar indices on the relative health of Fish Spawning Aggregations (FSAs) at the different sites through the examination of the different sizes of fish present. Further, this provided an additional case study to examine differences between using Single Echoes Detected (SED) and tracked fish as sources for the size classes.

6.3 Methods

6.3.1 Case studies

1. Cayman Islands.

The field surveys in Cayman occurred between the 14th and 20th of February 2014. Surveys were undertaken at fish spawning aggregation (FSA) sites of Nassau Grouper (*Epinephelus striatus*). These sites are all protected from fishing due to the presence of Designated Grouper Spawning Areas (DGSAs). The surveys utilised groundtruthing (towed video or SCUBA divers) to ensure that the aggregations were of the target species. Site nomenclature is as follows: Little Cayman West (LCW), Grand Cayman East (GCE), and Cayman Brac East (CBE). A number of different surveys were undertaken at each site and the number following the site code represents this in the figures and text. For more details on these surveys see Chapter 3 (Egerton et al. (2017)).

2. Qatar.

Surveys occurred from the 4th to 7th May 2015. Sites were chosen through examination of bathymetric charts and local knowledge and within each of these a $1km^2$ box was comprehensively surveyed by 8 parallel transects. All sites are open for fishing, although the degree of this at each site is unknown. There are however anecdotal reports of fishermen favouring the 'shallow' sites due to higher fish abundances (M. Abdallah, pers comm).

'Shallow' sites**:** These 7 sites aimed to target raised mounds known locally as hairãt (Smyth et al. 2016) and have a patchy distribution amongst the surrounding deeper waters with muddier sediments. These mounds, oyster beds or reefs are mainly located in water depths of 10-20m and throughout this manuscript we name them 'shallow' category sites. They have more consolidated coarse and rugose substratums resultant from the current or historic presence of oysters, corals and other reef forming organisms. Further, the site S6 (a coral reef) is placed in this category.

'Deep' sites: These 7 sites were located in the deeper waters surrounding the mounds are named 'deep' category sites. These sites have depths of circa 30-40m comprising finer and more mobile sediments of mud and sand. For more details see Chapter 5.

Table 6.1 The groundtruthed habitat types and depths at the 14 survey stations in the Qatari Gulf.

Site	Groundtruthed Habitat	Mean Depth (m)	±S.E.M
S1	Reef	13.47	0.038
S2	Sand	21.42	0.023
S3	Reef	18.95	0.035
S4	Reef	17.90	0.024
S5	Reef/Sand	18.96	0.014
S6	Reef inc live Coral	21.89	0.072
S7	Reef	16.94	0.016
D1	Mud	32.17	0.042
D2	Mud	32.46	0.016
D3	Mud	32.48	0.007
D4	Mud	27.58	0.003
D5	Mud	28.27	0.005
D6	Mud	31.70	0.007
D7	Mud/Sand	27.44	0.028

3. Cabo Pulmo National Park (CPNP), Baja California, Mexico.

The CPNP is composed of a mixture of habitats with basalt dikes, or reefs, forming long, parallel ridges that run adjacent to shore in the bay's northern section, while disappearing under the shoreline in the south-central section (Riegl et al. 2007). Between the reefs there are areas of sand and boulders. Controls areas were based at Punta Arena (PA) located 5 km to the North of the CPNP and Bajo del Salado (BS) 5 km to the South. Both areas experience fishing pressure and the seabed habitat (confirmed via towed camera) is dominated by sand at PA and rock and boulders at BS. The whole park is effectively managed as a no take by the local community (Jones, 2011). Unfortunately there is no quantitative information available on the

levels of fishing at the control sites although we did observe fishing at both locations. For more details see Chapter 4.

Table 6.2 The groundtruthed habitat types and protection regime at the survey stations in Mexico.

Site	Protection	Habitat
Reefs	✓	Rocky basalt reef with some coral (<20%)
Cabo Pulmo National Park (CPNP)	✓	Heterogeneous (Reefs as above, rocks, boulders and sand)
Bajo del Salado (BS)	x	Rocks and boulders
Punta Arena (PA)	x	Sand

6.3.2 Equipment

A Biosonics® DTX Split beam echosounder with a 200 kHz transducer was used for the surveys, pole mounted over the side of the survey vessel with the transducer face 1m under the surface. Acoustic data were georeferenced with an integrated Garmin 17Xhvs GPS, and collected with the Biosonics acquisition software (Visual Acquisition 6). The circular transducer has a beam-opening angle of 6.8°. Pulse duration was 0.4 ms and the specified ping rate was 10 per second. Calibration was performed using Tungsten Carbide 200 kHz Calibration Sphere before the surveys following the standard methods of Foote et al. (1987).

6.3.3 Acoustic data processing methods

Data were collected with the Biosonics software Visual Acquisition 6 as DT4 files and then converted and post-processed with the SONAR5 Pro (ver 6.0.4) software package (Balk and Lindem, 2006). Analysis in SONAR5 Pro followed the Software Guided Analysis (SGA) routine based upon the Standard Operating Procedure for the Great Lakes hydroacoustic surveys (see Parker-Stetter et al. 2009 for details). Abundance was determined through echo-integration rather than echo-counting as recommended by de Kerckhove et al. (2015) as echo-counting led to lower estimates of abundance and size-spectra indicators. Echo-integration (also known as Sv/TS scaling), divides the average reflection from all fish over a specified area (the volume backscattering coefficient, sv with units of $m^2\,m^{-3}$) by the average backscattering cross section (σ_{bs}) which in turn is derived from the mean echo intensity Target Strength (TS) from individual fish (Draštík et al. 2009; Winfield et al. 2011). Sonar5 (ver 6.0.4) applies a

Time Varied Gain (TVG) correction of 40log(R) for TS values and 20log(R) for Sv values (Balk and Lindem 2006). There are two options available to gain TS *in situ*, using Single Echoes Detected (SEDs) or using tracked fish; with a consecutive string of SEDs originating from a single fish. Analyses were based both by using Single Echo Detections (SED) and tracked fish as sources for TS and resultant data compared. The criteria applied to accept SEDs was a minimum echo length of 0.8 dB, a maximum of 1.2 dB and a maximum angle standard deviation of 0.8 degrees. Multi-peak suppression was set to 'medium' in the software which demands a local dip of 1.5 dB between peaks before rejecting the echo (Balk and Lindem 2006). With tracked fish there was a criteria of a minimum track length of 3 pings, a maximum ping gap of 2 pings, a gating range of 0.3 m applied to define a track. In the multispecies case studies (Qatar and Mexico), returns below a threshold of -60 dB were excluded to detect fish from other particulate targets such as plankton (Reid, 2000). With the Cayman dataset, TSs were restricted to those likely to be derived from Nassau Grouper (as groundtruthed by diver or underwater video). In order to ensure that no echoes from the seabed were classified as fish and the presence of the 'acoustic dead zone (Ona and Mitson, 1996), a bottom layer of 1m was applied and any returns from this layer were removed. A surface layer of around 1m (depending on the sea state) was also applied to the data to remove surface noise caused by waves. Whole transects were taken as Elementary Sampling Distance Units (EDSU's) to minimise the number of EDSUs with insufficient fish echoes for the algorithms (Balk and Lindem 2006). Similarly, transects were not divided into horizontal depth bins to minimise potential spatial autocorrelation, a problem that is avoided if Sv is calculated for the entire isonified water volume (Emmrich et al. 2010). Analyses therefore included both pelagic and demersal elements of the fish community. Mean target strengths were checked for bias (based on Yule et al. (2013) and Rudstam et al. (2009) following Sawada et al. (1993) ensuring number of fish per isonified volume (Nv) was less than 0.1.

To undertake hydroacoustic fish surveys, there needs to be adequate coverage over the survey areas to gain a reliable picture of the fish abundance. Degree of coverage (Λ) is defined as:

$$\Lambda = D/\sqrt{A}$$

where: D is the cruise track length; and, A is the size of the survey area (Aglen, 1989), and for adequate coverage the ratio needs to be 6 or over. This was achieved in all the different case study survey sites.

6.3.4 Fish size determination

Target Strength (TS) is used as an indicator of fish size but it varies between fish species (due to fish morphology, namely the size of the swim bladder) and also with the aspect of the fish. In order to convert TS into more intuitive length measurements (cm) than the decibel (Boswell et al. 2007), it is converted by using empirical TS-length relationships, which often exists for specific groups or species (Kracker, 2007). This was possible in the Cayman Case study where diver based measurements allowed derivation of a TS-Length formula for the Nassau Grouper (Chapter 3) (Egerton et al. 2017) to be applied:

TS = 27.6 log10(L)-147.32

At the case study sites in Qatar and Mexico, there was a wide diversity of fish species in the area making the use of species-specific TS–length formulas problematic (Coll et al. 2006). The multi-species equation from Love (1971) was therefore used to convert TS to total length. It provides the following conversion:

TS= (19.1 log10L) – (0.9 log$_{10}$ f) – 62.0

Where TS = target strength detected (dB), L = length of the target (cm) (Total Length), and f = the frequency used. As in this survey the transducer frequency was 200 kHz this equation then becomes:

TS=19.1 log$_{10}$(L) -64.07

Data based on SED were processed into 1dB bins as mean percentage frequencies of all transects per site, whilst data from tracked fish was processed into appropriate bin sizes (chosen to reduce the occurrence of zero values within a bin whilst allowing highest resolution possible). The midpoint of each bin was then converted from TS to length by the Love (1971) multi-species formula, and then from Length to weight using the formula for all fusiform fish (Froese, 2006): $\mathbf{W = a\ L^b}$

where W = weight (g), L= Total Length (TL) (cm). With constants **a = 0.0137** and **b =3.03.**

In the case study of the Cayman Islands a survey and species-specific TS-Length relationship was used to calculate length and the species-specific W-L formula with constants **a = 0.01122, b = 3.05** (Froese and Pauly 2016).

6.3.5 Size spectra analyses

The size classes and total fish biomass in each bin were then Log_{10} transformed. In the case of total fish biomass per size class this was a $Log_{10}(x+1)$ transformation so that '0' values were not removed through calculation error. Adding one to the count avoids bins with zero counts not appearing in the plots and not contributing to the regression calculation (Edwards et al. 2016). In order to normalise the data, the total weight of a given class was divided by the width of that size class. Vacant length classes were excluded from spectral regressions to eliminate a floor effect that may skew parameters (Rice and Gislason 1996; Wheeland and Rose, 2015).

Linear regression trendlines were then applied to the data and in this manner slopes, intercepts and midpoint heights were gained for each site. Quadratic trendlines were also fitted through the data and resultant R^2 and P values compared. Differences between intercepts, slopes and midpoint heights from the linear regression were examined between sites to see if differences in exploitation levels and/or habitat could be discerned. As intercept and slope are correlated (Daan et al. 2005; Sweeting, 2009; Gómez-Canchong et al. 2013) midpoint heights were plotted against slope for the different sites.

In the case of the Mexico dataset, it was not possible to test between transects due to high variability and many weight bins with no observations which made statistical analysis difficult (Sprules and Barth, 2015). To combat this we examined mean values from all transects within each site at all case studies. Statistical comparisons between mean slopes and midpoint heights per site were however possible with the Qatar dataset as there was sufficient repetition over the two different habitat categories. For all the case studies, statistical comparison (linear regression) was possible between slopes and midpoints derived from SED and from Tracked fish.

For all the case studies, how well the data fit the linear model is examined through R^2 values and associated P values through linear regression. We also investigate how well the data fit a domed structure through examination of R^2 and P values of quadratic equations through curvilinear regression. The R^2 value will always increase with addition of a higher-order term, but the question is whether the increase in R^2 is significantly greater than expected due to chance (McDonald, 2014). This is investigated through the P value associated with the increase in R^2.

6.4 Results

6.4.1 Case Study 1. Fish Spawning Aggregations (FSAs) in the Cayman Islands

NBSS are first calculated for tracked fish as the source of TS (Figure 6.2) and then for SEDs as the source for TS (Figure 6.4) for FSAs of Nassau Grouper in the Cayman Islands. The resultant slopes and midpoint heights are then shown for each data source (Figure 6.3and Figure 6.5). These data are then summarised in Table 6.3a and b, which also includes R^2 values for linear and quadratic functions, with associated values.

1a. Cayman Islands: Tracked fish.
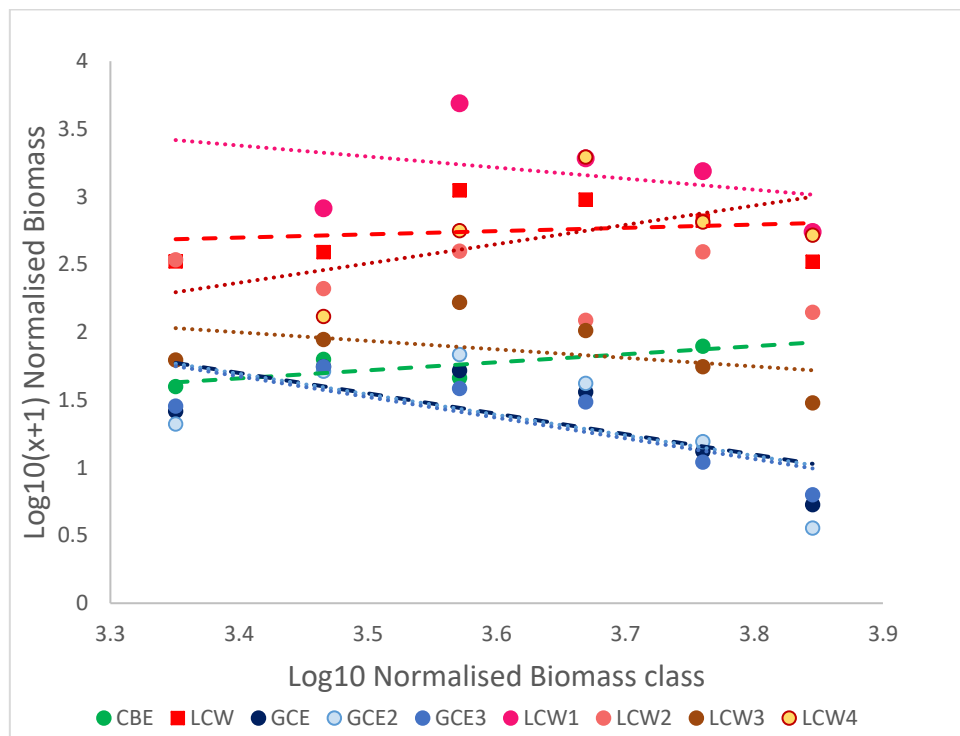
Figure 6.2. Linear trendlines of acoustic normalised biomass spectra, derived from Tracked fish showing mean values per site from surveys of Nassau Grouper FSAs in the Cayman Islands. See legend for survey site colour codes.

The slopes and midpoints from Figure 6.2 above are plotted per site below (Figure 6.3) to help summarise the data.
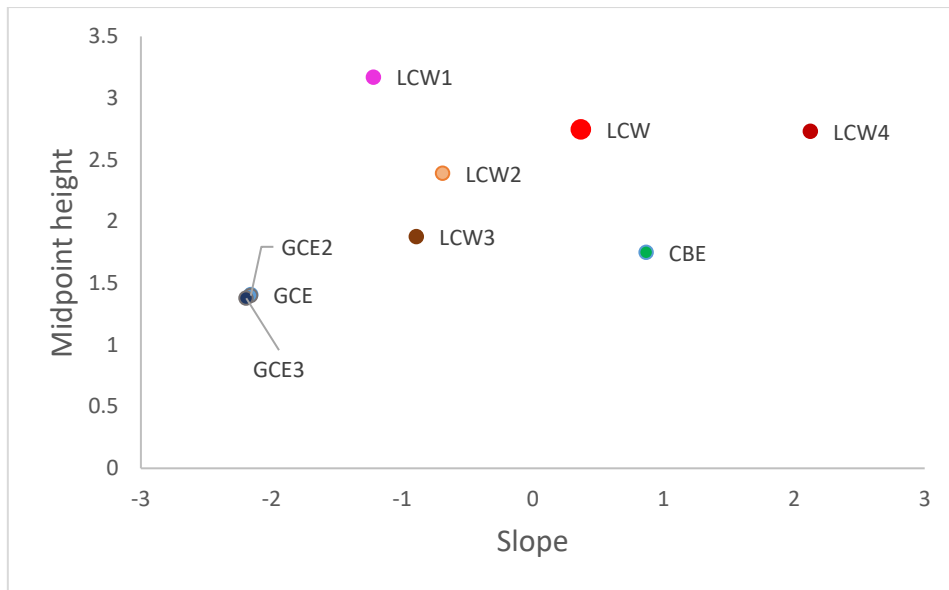
Figure 6.3. Tracked fish midpoint heights and slopes for spawning aggregations of Nassau Grouper in the Cayman Islands.

1b. Cayman Islands: SED
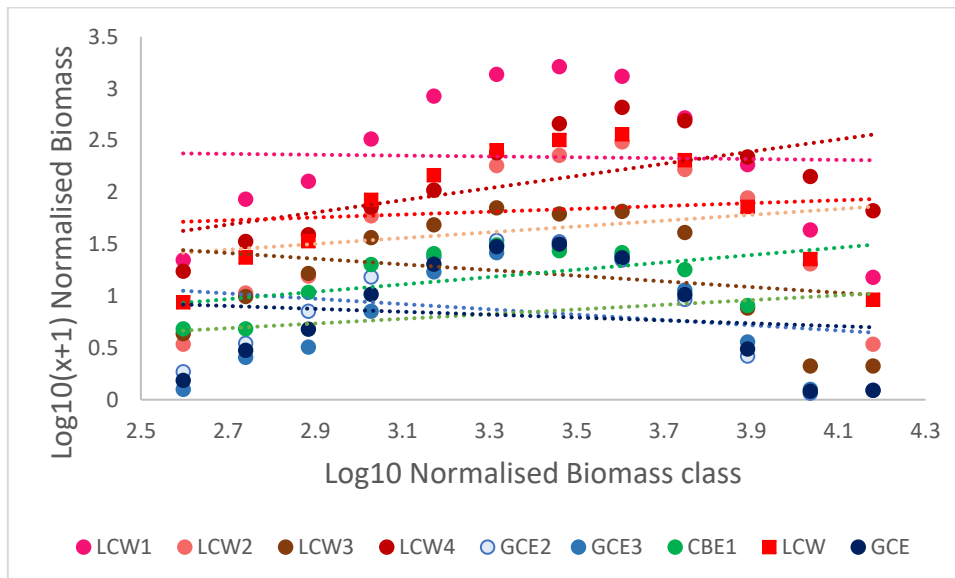
Figure 6.4. Linear trendlines of acoustic normalised biomass spectra, derived from SED showing mean values per site from surveys of Nassau Grouper FSAs in the Cayman Islands. See legend for survey site colour codes.
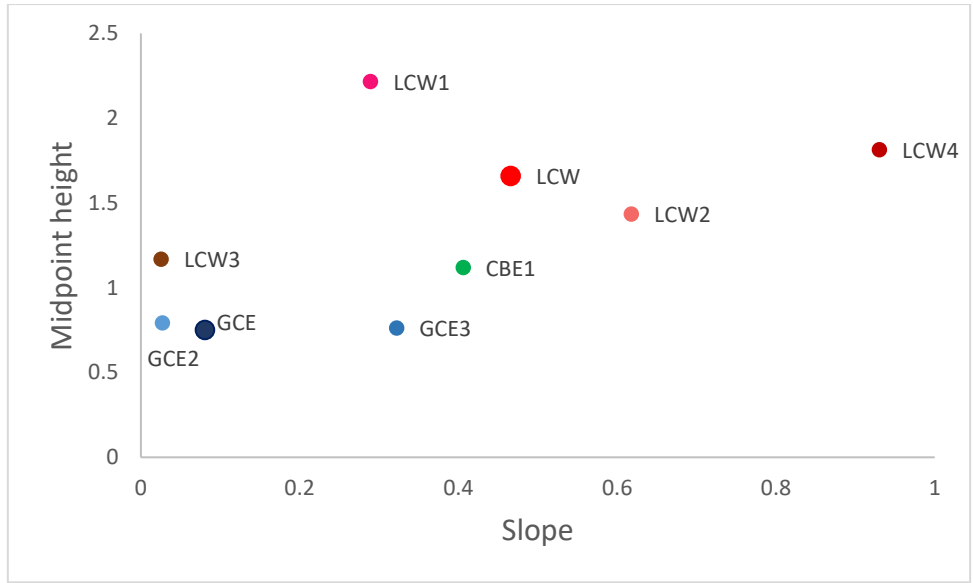
Figure 6.5. Midpoint heights and slopes (derived from SED) for spawning aggregations of Nassau Grouper in the Cayman Islands.

Slopes, intercepts and mid-point heights, R^2 values and P values associated with the linear versus quadratic fits for the Cayman Data are given in Table 6.3, below.

Table 6.3 Slopes, Intercepts and Midpoint heights for NBSS of Nassau Grouper aggregations in the Cayman Islands with SEDs (a) and tracked fish (b) as sources for TS. The R^2 values and associated P values for both linear and quadratic models are given. Also shown is the P value increase in R^2 with the quadratic model as opposed to the linear model.

a) Single Echoes Detected (SED)

Survey	Slope	Intercept	Midpoint height	R^2 linear	P-value	R^2 quadratic	P-value	P-value of increase in R^2
LCW1	0.290	1.303	2.216	0.059	0.425	0.620	**0.008**	**0.001**
LCW2	0.618	-0.514	1.434	0.234	0.515	0.941	**0.000**	**0.000**
LCW3	0.026	1.086	1.168	0.001	0.928	0.626	**0.007**	**0.001**
LCW4	0.931	-1.120	1.813	0.607	0.037	0.867	**0.000**	**0.000**
LCW	0.466	0.189	1.658	0.187	0.140	0.722	**0.002**	**0.000**
GCE2	0.027	0.707	0.793	0.001	0.921	0.616	**0.008**	**0.001**
GCE3	0.323	-0.256	0.761	0.133	0.244	0.405	0.097	0.054
GCE	0.081	0.495	0.750	0.009	0.761	0.540	**0.021**	**0.003**
CBE1	0.406	-0.162	1.118	0.437	**0.027**	0.585	0.030	0.115

b) Tracked Fish

Survey	Slope	Intercept	Midpoint height	R^2 linear	P-value	R^2 quadratic	P-value	P-value of increase in R^2
LCW1	-0.814	6.147	3.169	0.112	0.582	0.787	0.213	0.136
LCW2	-0.469	4.075	2.387	0.145	0.457	0.147	0.787	0.991
LCW3	-0.628	4.135	1.874	0.209	0.363	0.924	**0.021**	**0.011**
LCW4	1.423	-2.474	2.729	0.260	0.380	0.871	0.129	0.095
LCW	0.240	1.882	2.746	0.036	0.720	0.776	0.106	**0.048**
GCE2	-1.513	6.835	1.393	0.358	0.210	0.998	**0.000**	**0.000**
GCE3	-1.524	6.856	1.372	0.624	0.062	0.959	**0.008**	**0.014**
GCE	-1.505	6.815	1.401	0.501	0.115	0.991	**0.001**	**0.001**
CBE1	0.594	-0.361	1.752	0.596	0.228	0.601	0.632	0.989

There was a significant relationship between slopes derived from SED and slopes derived from tracked fish ($R^2 = 0.626$, $P_8 = 0.011$). This was also the case in examining midpoint heights from the two methods ($R^2 = 0.979$, $P_8 < 0.01$).

One would expect LCW to have the least negative/most positive slope rather than CBE, however there were few fish recorded at CBE and zero values within the size bins had to be removed to avoid the 'floor effect' as described by Rice and Gilason, (1996). It is also interesting that slopes for Tracked fish tend to be negative whereas those from SED positive.

Examination of the linear P values and associated R^2 values were not significant for all spectra except CBE based on SED, meaning that these slopes should be treated with caution. The limited size range of fish in the aggregations results in significantly better fits with quadratic models, especially when using SED. The midpoint heights of the spectra are seen to be higher at LCW than the other sites, which showing greater fish abundance. Midpoint heights gained from using Tracked fish were higher at all sites than that for SED.

6.4.2 Case Study 2. Fish between different habitat types in Qatar

NBSS are first calculated for tracked fish as the source of TS of site means (Figure 6.6) and then with means of each habitat category (Figure 6.7). NBSS for SEDs as the source for TS per site (Figure 6.9) and per habitat category (Figure 6.10) for the multispecies fish community between habitat types in the Qatari Gulf. The resultant slopes and midpoint heights are then shown for each data source (Figure 6.8 and Figure 6.11). These data are then summarised in Table 6.4a and b, which also includes R^2 values for linear and quadratic functions, with associated P values.

2a. Qatar: Tracked fish
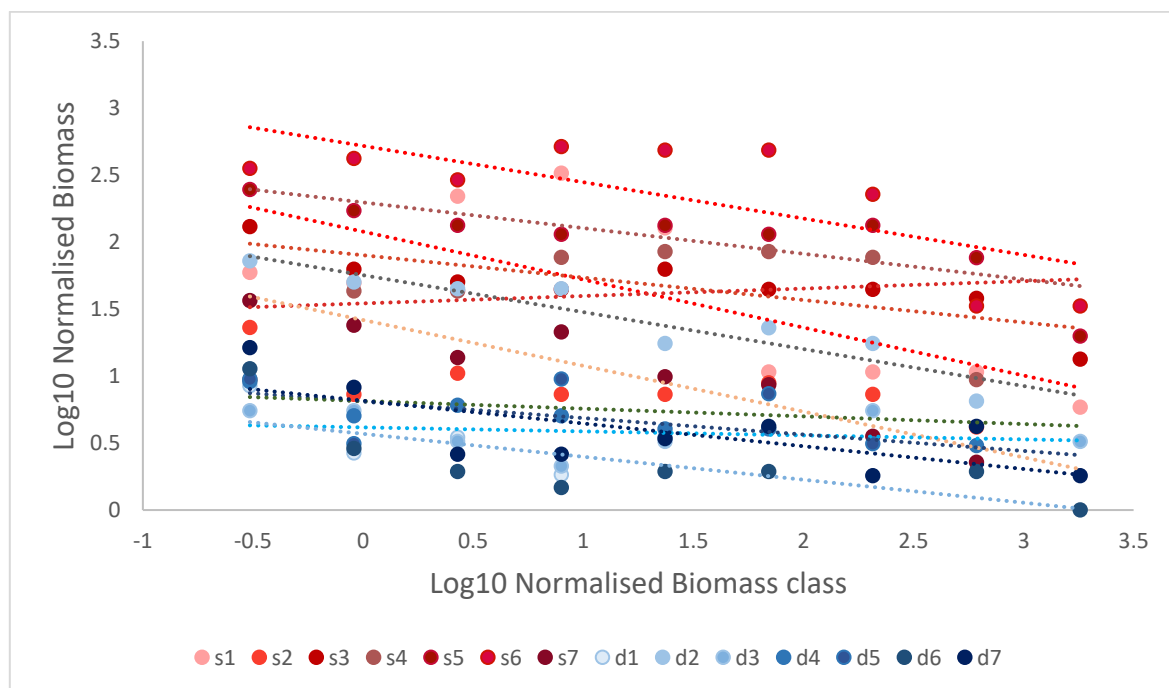
Figure 6.6. Acoustic normalised biomass spectra with linear trendlines, derived from Tracked fish showing mean values per site from surveys in Qatari Gulf. 's' represents 'shallow' sites (oysterbed/reef) which are shown in red, 'd' represents 'deep' sites (sand/mud habitat), shown in blue.
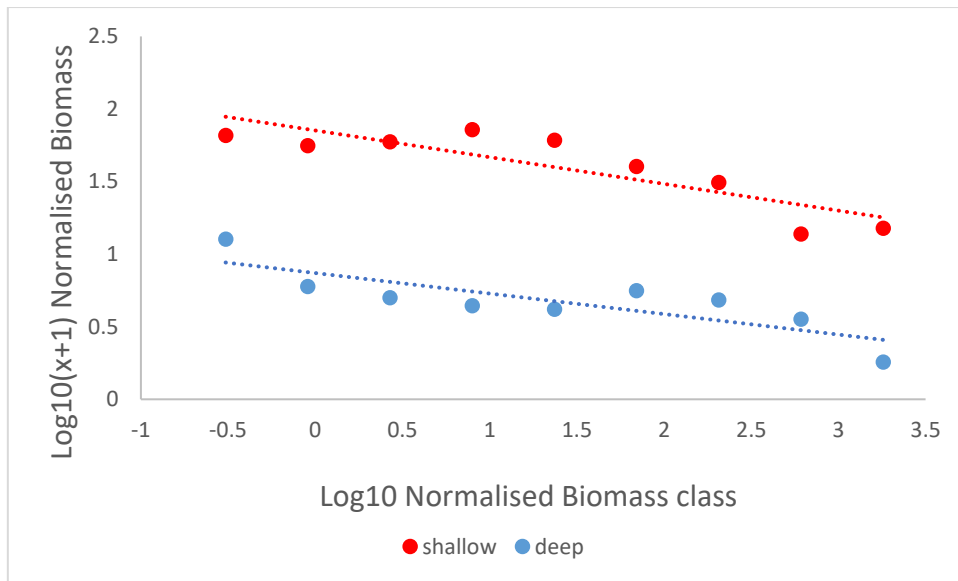
Figure 6.7. Acoustic normalised biomass spectra, derived from single echoes detected (SED) from surveys mean values per habitat category in Qatari Gulf. 'shallow' sites (oysterbed/reef) are shown in red whilst 'deep' sites (sand/mud habitat)are shown in blue.
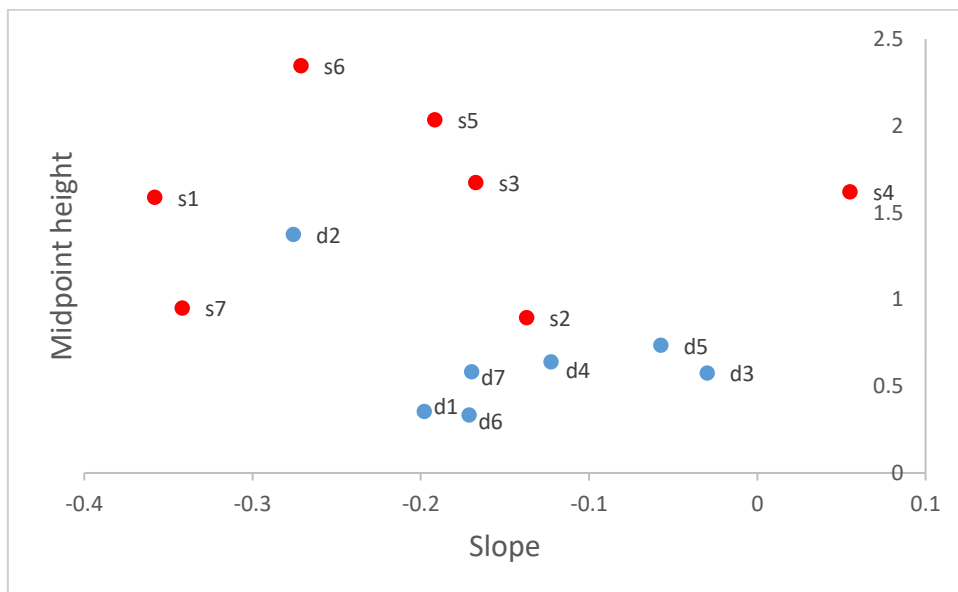
Figure 6.8. Mean size spectra slopes plotted against mean size spectra midpoint heights derived from Tracked fish for deep vs shallow survey sites in Qatar. 'shallow' sites (oysterbed/reef) are shown in red whilst 'deep' sites (sand/mud habitat)are shown in blue.

2b. Qatar: SED
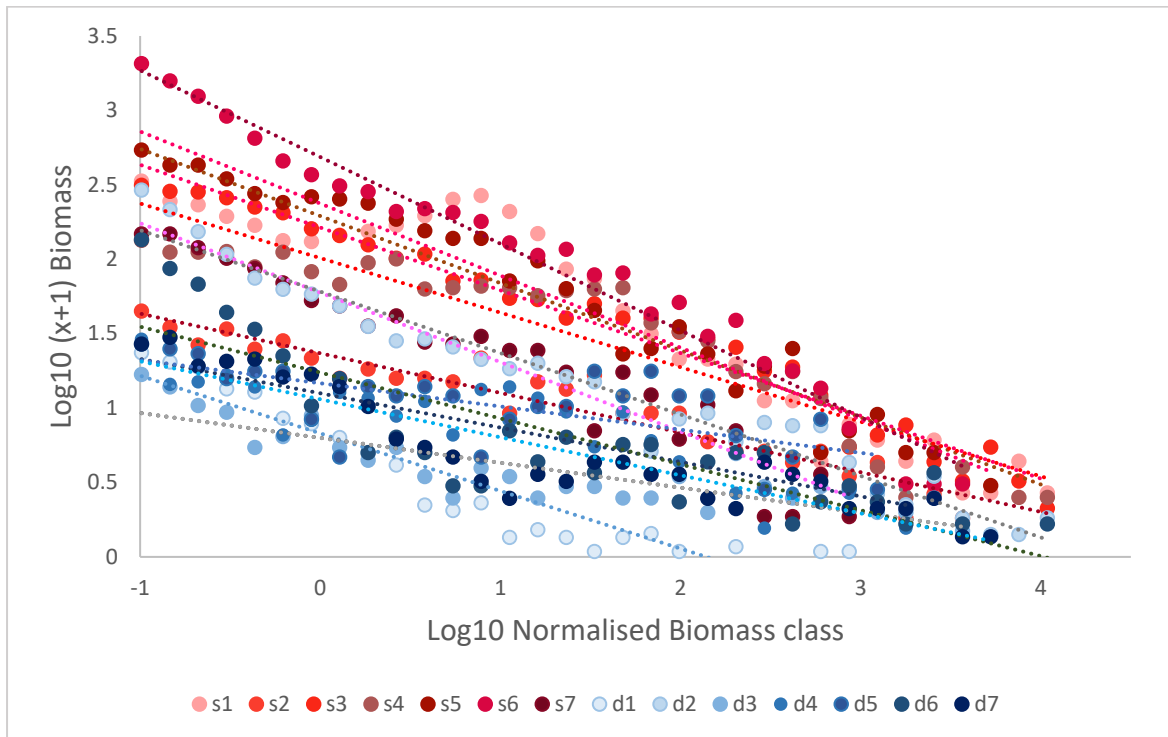
Figure 6.9. Acoustic normalised biomass spectra, derived from single echoes detected (SED) showing mean values per site from surveys in Qatari Gulf. 's' represents 'shallow' sites (oysterbed/reef) which are shown in red, 'd' represents 'deep' sites (sand/mud habitat), shown in blue.
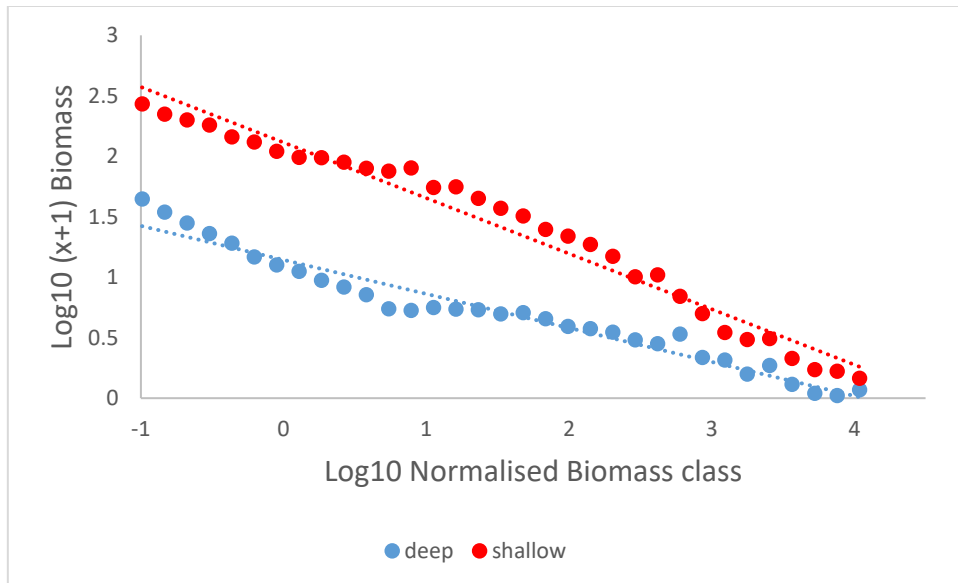
Figure 6.10. Acoustic normalised biomass spectra, derived from single echoes detected (SED) from surveys mean values per habitat category in Qatari Gulf. 'shallow' sites (oysterbed/reef) are shown in red whilst 'deep' sites (sand/mud habitat)are shown in blue.
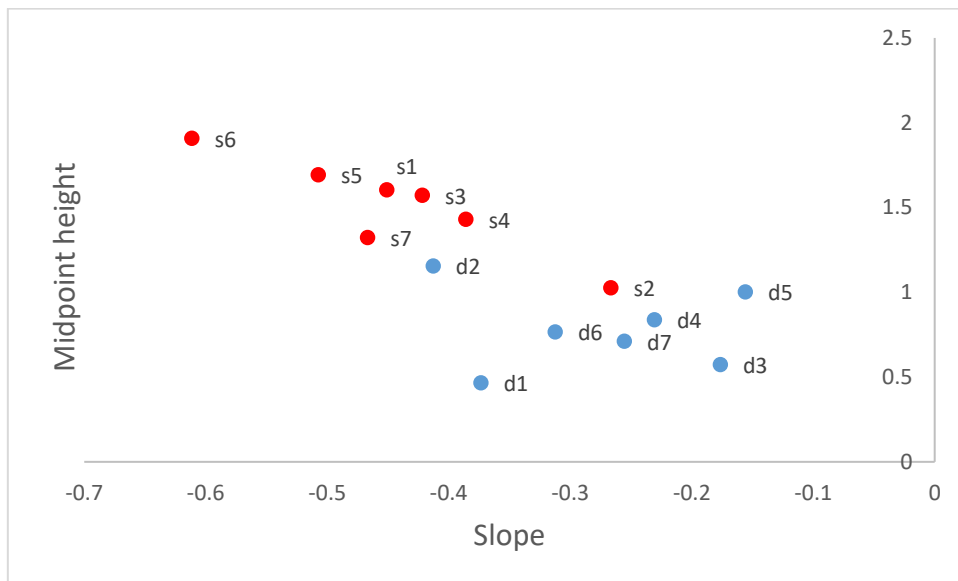
Figure 6.11. Mean size spectra slopes plotted against mean size spectra midpoint heights (derived from SED) for deep vs shallow survey sites in Qatar. 'shallow' sites (oysterbed/reef) are shown in red whilst 'deep' sites (sand/mud habitat)are shown in blue.

Table 6.4 Slopes, Intercepts and Midpoint heights for NBSS of the Qatar dataset, with SEDs (a) and tracked fish (b) as sources for TS. The R^2 values and associated P values for both linear and quadratic models are given. Also shown is the P value increase in R^2 with the quadratic model as opposed to the linear model.

a) SED

Site	Slope	Intercept	Midpoint height	R^2 linear	P-value	R^2 quadratic	P-value	P-value of increase in R^2
s1	-0.451	2.290269	1.604	0.888	**0.000**	0.927	**0.000**	**0.000**
s2	-0.266	1.368277	1.026	0.921	**0.000**	0.940	**0.000**	**0.002**
s3	-0.422	2.212932	1.571	0.972	**0.000**	0.982	**0.000**	**0.000**
s4	-0.386	2.01762	1.430	0.857	**0.000**	0.950	**0.000**	**0.000**
s5	-0.507	2.384989	1.692	0.945	**0.000**	0.968	**0.000**	**0.004**
s6	-0.611	2.694312	1.908	0.947	**0.000**	0.981	**0.000**	**0.001**
s7	-0.467	1.775538	1.322	0.935	**0.000**	0.942	**0.000**	0.075
d1	-0.374	0.828922	0.466	0.878	**0.000**	0.976	**0.000**	**0.000**
d2	-0.413	1.782635	1.154	0.960	**0.000**	0.967	**0.000**	**0.005**
d3	-0.177	0.801249	0.574	0.735	**0.000**	0.829	**0.000**	**0.000**
d4	-0.231	1.099352	0.839	0.777	**0.000**	0.813	**0.000**	**0.016**
d5	-0.156	1.166756	1.003	0.467	**0.000**	0.475	**0.000**	0.710
d6	-0.312	1.240766	0.765	0.756	**0.000**	0.851	**0.000**	**0.000**
d7	-0.255	1.059154	0.711	0.837	**0.000**	0.897	**0.000**	**0.000**
shallow	-0.397	2.087028	1.701	0.966	**0.000**	0.993	**0.000**	**0.000**
deep	-0.284	1.144326	0.868	0.918	**0.000**	0.965	**0.000**	**0.001**

b) Tracked Fish

Site	Slope	Intercept	Midpoint height	R^2 linear	P-value	R^2 quadratic	P-value	P-value of increase in R^2
s1	-0.358	2.078	1.587	0.511	**0.031**	0.676	**0.034**	0.122
s2	-0.137	1.081	0.893	0.564	**0.032**	0.586	0.110	0.782
s3	-0.167	1.902	1.673	0.691	**0.005**	0.704	**0.026**	0.788
s4	0.055	1.542	1.618	0.024	0.712	0.836	**0.011**	**0.003**
s5	-0.192	2.296	2.033	0.647	**0.009**	0.745	**0.017**	0.182
s6	-0.271	2.718	2.346	0.528	0.027	0.854	**0.003**	**0.006**
s7	-0.342	1.418	0.950	0.907	**0.000**	0.937	**0.001**	0.184
d1	-0.198	0.625	0.354	0.361	0.284	0.766	0.234	0.224
d2	-0.276	1.753	1.375	0.872	**0.001**	0.897	**0.003**	0.365
d3	-0.030	0.617	0.576	0.061	0.593	0.259	0.549	0.424
d4	-0.123	0.809	0.641	0.838	**0.004**	0.854	**0.021**	0.685
d5	-0.057	0.814	0.735	0.077	0.651	0.164	0.836	0.828
d6	-0.171	0.569	0.334	0.532	0.123	0.815	**0.034**	**0.034**
d7	-0.170	0.816	0.583	0.493	**0.035**	0.645	**0.045**	0.156
shallow	-0.184	1.852	1.600	0.747	**0.003**	0.912	**0.001**	**0.009**
deep	-0.141	0.870	0.676	0.673	**0.007**	0.686	**0.031**	0.728

Slopes at shallow sites were significantly different from those at deep with SED sites (T_{11} = 3.13, P = 0.010) but not tracked fish (T_9 = 0.89, P = 0.398) with steeper slopes present over the more rugose shallow sites. Midpoint heights were significantly higher at the shallow category sites for both SED (T_{11} = 5.16, P < 0.001) and Tracked fish (T_{10} = -3.88, P = 0.003). Site S6 had the highest midpoint height for both SED (2.694) and Tracked fish (2.346). Using SED, there were generally significantly better fits with using quadratic models, the only insignificant improvements seen with sites s7 (P = 0.075) and d5 (P = 0.71). There was a significant relationship between slopes derived from SED and slopes derived from tracked fish (R^2=0.34, P_{15} = 0.011). This was also the case in examining midpoint heights from the two methods (R^2 = 0.89, P_{15} <0.01). Generally however, there were higher midpoints revealed using SED rather

than using tracked fish. Steeper slopes were seen at all sites with the SED dataset. This is likely to be due to it being more difficult to gain valid fish tracks at the smaller end of the size scale.

6.4.3 Case Study 3. The effects of marine protection and different habitat types in Mexico

NBSS are calculated for tracked fish as the source of TS (Figure 6.12) and then for SEDs as the source for TS (Figure 6.14) for the multispecies community at the CPNP, the reefs within it, and also at control sites (PA and BS), using mean values per site. The resultant slopes and midpoint heights are then shown for each data source (Figure 6.13and Figure 6.15). These data are then summarised in Table 6.5a and b, which also includes R^2 values for linear and quadratic functions, with associated P values.

3a. Mexico: Tracked fish

Figure 6.12. Acoustic normalised biomass spectra, derived from Tracked fish showing mean values per site from surveys in Mexico. Site nomenclature as shown in Table 6.2.

Figure 6.13. Mean size spectra slopes plotted against mean size spectra midpoint heights, derived from Tracked fish, for the different sites investigated. Site nomenclature as shown in Table 5.2.

3b. Mexico: SED
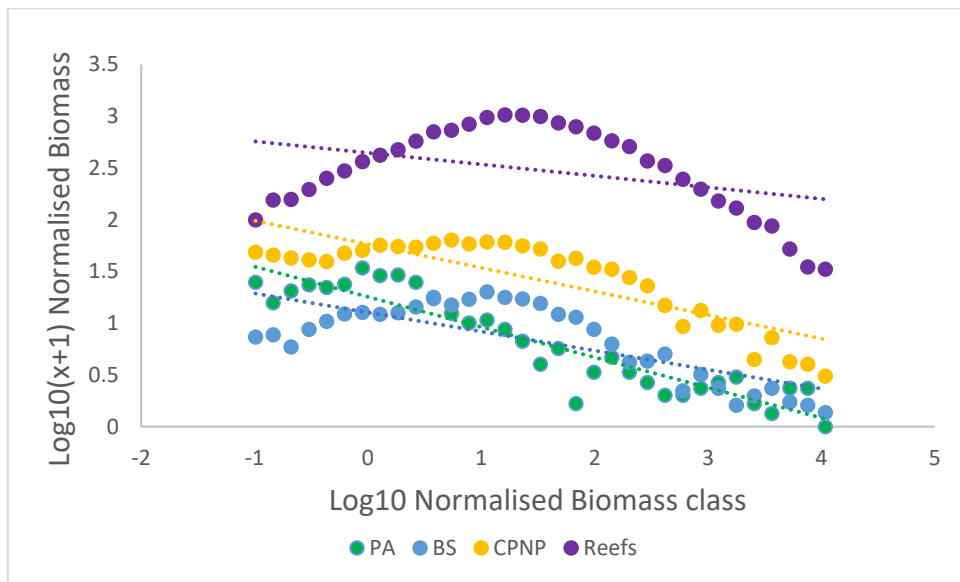
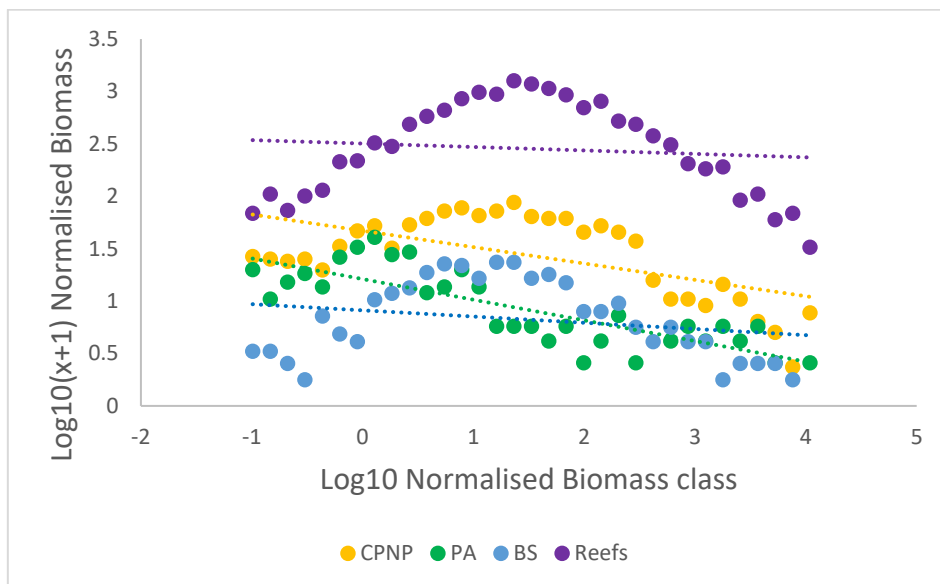*Figure 6.14. Acoustic normalised biomass spectra, derived from SED showing mean values per site from surveys in Mexico. Site nomenclature as shown in Table 6.2.*
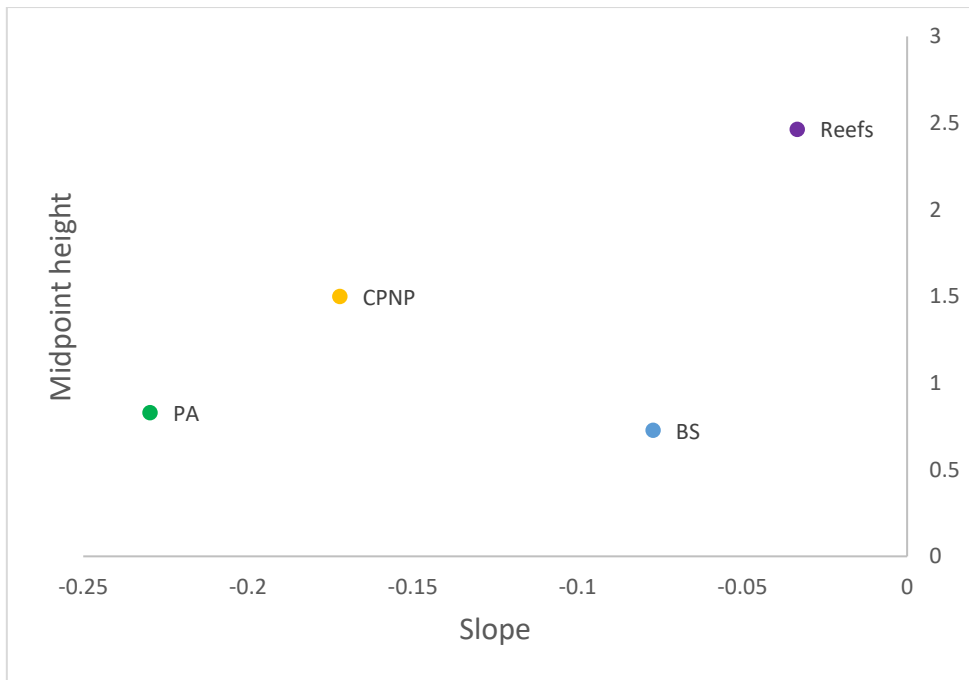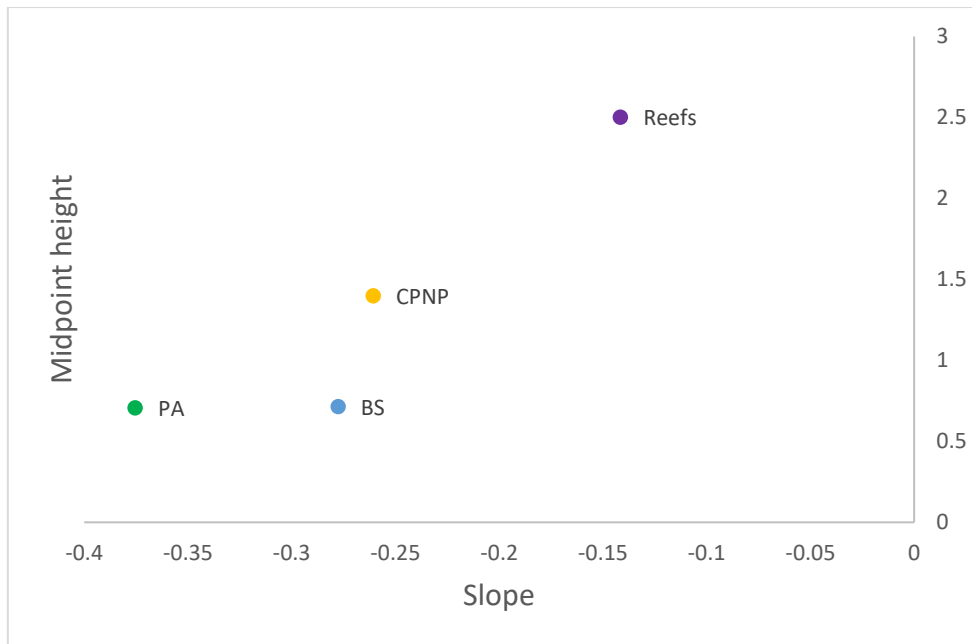
Figure 6.15. Mean size spectra slopes plotted against mean size spectra midpoint heights, derived from Tracked fish, for the different sites investigated. Site nomenclature as shown in Table 6.2.

Table 6.5 Slopes, Intercepts and Midpoint heights for NBSS of the Mexico dataset, with SEDs (a) and tracked fish (b) as sources for TS. The R^2 values and associated P values for both linear and quadratic models are given. Also shown is the P value increase in R^2 with the quadratic model as opposed to the linear model.

a) Single Echoes Detected (SED)

Site	Slope	Intercept	Midpoint height	R^2 linear	P-value	R^2 quadratic	P-value	P-value of increase in R^2
PA	-0.375	1.248	0.706	0.847	0.000	0.852	**0.000**	0.910
BS	-0.278	1.138	0.715	0.607	0.000	0.884	**0.000**	**0.000**
CPNP	-0.261	1.793	1.397	0.719	0.000	0.964	**0.000**	**0.000**
Reefs	-0.142	2.716	2.501	0.230	0.025	0.983	**0.000**	**0.000**

b)Tracked Fish

Site	Slope	Intercept	Midpoint height	R^2 linear	P-value	R^2 quadratic	P-value	P-value of increase in R^2
PA	-0.197	0.910	1.210	0.653	**0.000**	0.653	**0.000**	0.986
BS	-0.059	0.836	0.912	0.055	0.195	0.788	**0.000**	**0.000**
CPNP	-0.156	1.519	1.671	0.347	**0.000**	0.857	**0.000**	**0.000**
Reefs	-0.033	2.466	2.503	0.012	0.540	0.946	**0.000**	**0.000**

The site with greatest fish biomass (Reefs) has the highest midpoint height and lowest slope, whereas the site with the lowest biomass (PA) with the lowest midpoint and highest slope, separates furthest from Reefs. The CPNP and BS are located relatively close together, but CPNP has a higher midpoint, indicating higher abundance, and BS has a steeper slope potentially due to size selective fishing activity having removed a proportion of larger fish at this site.

There was not a significant relationship between slopes derived from SED and slopes derived from tracked fish (R^2= 0.67, P_3= 0.184). There was however a significant relationship between midpoint heights from the two methods (R^2= 0.97, P_3= 0.017). As with the Cayman dataset, this is potentially due to poor fits to linear models especially with the data based on tracked fish.

158

For both SED and tracked fish, the steepest slope was located at PA (no protection, sandy habitat) whilst the Reef transects (protection, favourable reef habitat) displayed the shallowest slope. The control sites at PA and BS had the smallest intercepts and midpoints whilst both of these parameters were highest over the Reef transects followed by the CPNP transects.

For all the sites, with both tracked fish and SED, the size spectra fitted quadratic models better than linear models. (Table 6.1). The Linear model however also fit well with PA (SEDs: R^2=0.847, P<0.001, Tracked fish: R^2=0.653, P<0.001), whereas Reefs show a relatively poor fit for this (SEDs: R^2=0.230, P=0.025, Tracked fish: R^2= 0.012, P=0.54). Conversely, Reefs have the closest fit with the quadratic model (SEDs: R^2=0.983, P<0.001, Tracked fish: R^2=0.946, P<0.001) whereas PA has the worst (SEDs: R^2=0.852, P<0.001, Tracked fish: R^2= 0.653, P<0.001). The only site which did not have a significant increase in R^2 values from linear models to applying quadratic ones was PA for both SED and Tracked fish (P = 0.910, P=0.986). This may reveal the effect of size-selective fishing and/or that the habitat type is more favourable for smaller fish at this site.

6.5 Discussion

6.5.1 SEDs vs Tracked Fish

In processing hydroacoustic data there are two methods of deriving *in situ* Target Strength (TS), the first is using Single Echoes Detected (SED) and the second is to take mean TS from fish along a consecutive string of SED, known as a fish track (Balk and Lindem, 2006) (See methods for further details). Both methods have been used to derive acoustic size spectra; SED by Yurista et al. (2014) and Tracks by Pollom and Rose, (2015) and de Kerckhove et al. (2015). None of these studies have however compared the size spectra resulting from the two different methods.

Size spectra approaches are routinely conducted to represent fish sizes across the different species within an ecosystem (Blanchard et al. 2017). Such studies on single species spectra are limited but have been examined theoretically (Andersen and Beyer, 2006; Law et al. 2012). In conducting such approaches with the Cayman Islands dataset, only one species is present and furthermore, the fish examined are from a narrow range of sizes, and all at a similar reproductive stage. Whilst such data can be clearly plotted as a histogram, size spectra were applied to help determine the differences between using SED and tracked fish as the source of

TS. The significant relationship between the resultant slopes and midpoints from the two datasets, adds confidence that the two methods are valid and potentially interchangeable with caution.

The patterns shown by application of both methods in our case studies are broadly similar. In Qatar and Cayman there was not a significant difference in slopes or heights between using SED or via tracked fish. In Mexico slopes were significantly different between methods but not heights. It may be the case that SED provides a more robust relationship due to both greater resolution through smaller bin sizes and less empty bins. The smoothed frequency distribution of SEDs may be partly caused in part by variable TS records obtained from fish being tilted off the horizontal axis (Emmrich et al. 2010). More reliable fish TSs should be present through the application of tracked fish (Ehrenberg and Torkelson, 1996), so the most suitable method is therefore likely to depend on the nature of the dataset and the number of accepted fish tracks available. However, as it can be seen that SEDs allow a broader size spectrum, largely as fewer fish are tracked in the smaller size classes. It is possible that some bias is introduced due to difficulties in extracting SEDs and fish tracks when the fish are highly aggregated, but it is difficult to discern how this may manifest itself in the data and more research into this area is recommended. Even in the case of the Cayman FSA dataset however, it was possible to extract enough fish echoes to enable logical NBSS. We acknowledge that our use of the multispecies equation from Love (1971) and conversion to weight (Froese, 2006) are unlikely to be appropriate for all the species of fish present when this was used in our Qatar and Mexico case studies. It does however provide a relative and consistent scale to compare between sites (Boswell et al. 2010; Pollom and Rose, 2015; Wheeland and Rose, 2015). Discerning individual echoes to specific species is always a challenge for acoustic fish surveys in a multispecies situation (Mackinson et al. 2004). There would be much merit in conducting further research comparing size spectra from the use of the multispecies equations with more species-specific formula when there is sufficient groundtruthing to allow this.

6.5.2 Quadratic vs linear models

Whether the size spectrum of marine fish communities is better described by a linear or a quadratic relationship remains an open question (Shin and Cury, 2004). Domes (quadratic relationships) were clear with the single species dataset from Cayman. This was as expected due to the limited size range of individuals who do not prey on one another. In fitting models

through the Mexican case study data, all sites were better described by quadratic models compared to linear models, although not significantly better at PA. The degree of curvature, and fit, of the quadratic line became less pronounced in the order of Reefs, CPNP, BS and finally PA with linear model showing the opposite trend. The extreme dome shaped of the NBSS over the reefs may be indicative of a relatively more pristine environment exhibiting a pattern towards that of an inverted trophic pyramid (Sandin et al. 2008; Mourier et al. 2016). The high proportions of larger fish over these reefs has also been noted by Aburto-Oropeza et al (2011) who found that since the creation of the park, abundances of top predators increased by 11 times and carnivores by 4 times. Initially our findings appear contrary to those other authors' who suggest that the addition of fishing pressure turns a linear relationship to a quadratic one (Rochet et al. 1999, Benoit and Rochet, 2004, Shin and Curry 2004). However, these studies do not include small fish (due to gear sampling limitations) and when small fish are excluded from size spectra a linear relationships result prevails (Rice and Gilason, 1996, Shin and Cury, 2004). In a quadratic relationship or dome, however, there are two sides to the parabola. Relatively low numbers of small fish will pull down the left side of the parabola, increasing curvature as noted by other authors (e.g. Bianchi et al. 2000; Ackerman et al. 2000), as will fewer large fish, caused by fishing, on the right hand side. Without the smaller fish shown by acoustics, we can see how fishing would increase curvature but with small fish data included as here (down to 2cm) this is not as straightforward. Few small fish may however also be resultant of these fish being elsewhere than the survey area (e.g. nursery areas). Other empirical studies where small fish are sampled also report more curvilinear patterns (Boudreau et al. 1991; Ackerman and Bellwood 2000; Munyandorero, 2006; Zgliczynski and Sandin, 2017). Sprules and Barth, (2015), using an optical plankton counter (with no size selectivity in the sampling) also found an abundance spectrum for zooplankton is better described by a quadratic model than a linear one. Further when modelling size spectra, dome-shaped biomass distributions are also revealed (Jacobsen et al. 2014; Engelhard et al. 2014; Anderson et al. 2016). Dome-shaped patterns in the spectra using models are much more marked than in survey data due to smaller fish being 'sampled' (Rice and Gislason, 1996). As our acoustic data allowed the inclusion of these smaller fish, a more complete description is possible; relatively low numbers in the small fish classes have an effect of increasing curvature, as do fewer large fish. With the Qatar dataset, quadratic models also fitted more rugose 'shallow' habitats better than the low rugosity deeper sites. More complex habitats have been seen to exhibit greater nonlinearities than low complex ones (Rogers et al. 2014). It is likely that a similar effect is

seen in present data, with greater curvature being present at more complex sites in both the Mexico and Qatar case studies. If, as found here, domes are present in acoustic marine NBSS, it may be difficult to use these in order to determine changes in the fish community (Shin et al. 2005). However, it is likely that other spectral-based indices can be developed to encompass this. For example, potentially by using the linear element on the descending limb of the size spectra (Munyandorero, 2006).

6.5.3 Potential effects of protection and habitat on NBSS slopes and heights

Whilst better fits were present with quadratic models, the indices of the spectra from linear models (slopes and heights) were still examined. In the Cabo Pulmo case study, when slopes plotted against midpoint heights from SEDs (discussed here due to better linear fits), Reefs can be seen to separate with highest values of both slope and height, followed by the CPNP, BS and lastly PA. These trends are likely be due to the combination of habitat differences and the effects of large fish removal at the fished control sites PA and BS (Thesis Hypothesis 6a). This follows previous findings where lower fishing pressure results in higher midpoint heights (e.g. Dulvy et al. 2004, Sweeting et al. 2009) and shallower slopes (Pope and Knights 1982; Blanchard et al. 2009; Robinson et al. 2016). The Qatar dataset provides better linear fits of NBSS than seen in Mexico. We hypothesise that this may be due to the healthier ecosystems present in the CPNP with greater proportions of larger fish increasing the curvature. In Qatar, we found significantly steeper slopes associated with more complex habitats, and the steepest slope (at site S6) is known to have the most complex habitat of these sites. We also see significantly higher spectral heights, a measure of abundance, over the more complex habitats at both of these case studies (Thesis Hypothesis 6b), which is in agreement with Wilson et al. (2010) and Alvarez-Filip et al. (2011). This however leads to an interesting difference in the NBSS patterns seen when slopes are plotted against midpoint heights between the Mexico and Qatar datasets. In Mexico, highest midpoints were associated with shallowest slope whereas in Qatar highest midpoints were associated with steepest slopes. This is likely to be resultant from complicated interplay between habitat and fishing pressure on the fish community. For example, this may occur due to the size-selective fishing occurring at the rugose 'shallow' sites in Qatar removing large fish and steepening the slope, whereas the protection offered by the rugose reefs in Mexico results in a greater abundance of large fish, shallowing the slope. More research into this with greater replication of sites with known fishing pressures and habitat types may reveal important considerations for marine managers.

Further testing of hydroacoustic methods, with their ability to sample fish throughout the size spectrum may show quadratic models demonstrate better fits than linear models as has been seen here. This is an important finding in interpreting acoustically-derived fish spectra data and may help further the understanding of spectra in general. Examination of midpoint heights and slopes using linear models has also led to agreement with other studies on fish spectra using other methods with respect to protection and habitat. Hydroacoustics could be used to examine pressing scenarios such as whether natural communities have a size-structured equilibrium state to which they return after disturbance (Murawski and Idoine, 1992; Andersen et al. 2016). Such methods could also be used to rapidly assess or monitor fish populations in an ecosystem approach context in response to anthropogenic (or environmental) changes (Jennings, 2005; Shin et al. 2005). Furthermore, the technique may reveal interesting trends on temporal size-based distribution between years, seasons and different times of day.

6.6 References

Aburto-Oropeza, O., Erisman, B., Galland, G.R., Mascareñas-Osorio, I., Sala, E. and Ezcurra, E., 2011. Large recovery of fish biomass in a no-take marine reserve. *PLoS One*, *6*(8), p.e23601.

Ackerman, J.L. and Bellwood, D.R., 2000. Reef fish assemblages: a re-evaluation using enclosed rotenone stations. *Marine Ecology Progress Series*, *206*, pp.227-237.

Aglen, A. 1989. Empirical results on precision-effort relationships for acoustic surveys. ICES CM: 1989/B:30, 28 p.

Alvarez-Filip, L., Reyes-Bonilla, H. and Calderon-Aguilera, L.E., 2006. Community structure of fishes in Cabo Pulmo reef, Gulf of California. *Marine Ecology*, *27*(3), pp.253-262.

Andersen, K.H. and Beyer, J.E., 2006. Asymptotic size determines species abundance in the marine size spectrum. *The American Naturalist*, 168(1), pp.54-61.

Andersen, K.H., Jacobsen, N.S. and Farnsworth, K.D., 2015. The theoretical foundations for size spectrum models of fish communities. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.575-588.

Balk, H. and Lindem, T. 2006. Sonar 4, Sonar 5, Sonar 6 e Post-processing Systems. Operator Manual. Lindem Data Acquisition, Oslo.

Benoît, E., and Rochet, M. J. 2004. A continuous model of biomass size spectra governed by predation, and the effects of fishing on them. *Journal of Theoretical Biology*, *226*: 9-21

Bianchi, G., Gislason, H., Graham, K., Hill, L., Jin, X., Koranteng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F. and Zwanenburg, K., 2000. Impact of fishing on size composition and diversity of demersal fish communities. *ICES Journal of Marine Science*, *57*(3), pp.558-571.

Blanchard, J.L., Jennings, S., Law, R., Castle, M.D., McCloghrie, P., Rochet, M.J. and Benoît, E., 2009. How does abundance scale with body size in coupled size-structured food webs?. *Journal of Animal Ecology*, *78*(1), pp.270-280.

Blanchard, J.L., Heneghan, R.F., Everett, J.D., Trebilco, R. and Richardson, A.J., 2017. From bacteria to whales: using functional size spectra to model marine ecosystems. *Trends in ecology and evolution*. 32(8) pp.174-186

Boswell, K.M., Wilson, M.P. and Wilson, C.A., 2007. Hydroacoustics as a tool for assessing fish biomass and size distribution associated with discrete shallow water estuarine habitats in Louisiana. *Estuaries and Coasts*, *30*(4), pp.607-617.

Boswell, K.M., Wells, R.J., Cowan Jr, J.H. and Wilson, C.A., 2010. Biomass, density, and size distributions of fishes associated with a large-scale artificial reef complex in the Gulf of Mexico. *Bulletin of Marine Science*, *86*(4), pp.879-889.

Boudreau, P.R. and Dickie, L.M., 1992. Biomass spectra of aquatic ecosystems in relation to fisheries yield. *Canadian Journal of Fisheries and Aquatic Sciences*, *49*(8), pp.1528-1538.

Boudreau, P.R., Dickie, L.M. and Kerr, S.R., 1991. Body-size spectra of production and biomass as system-level indicators of ecological dynamics. *Journal of Theoretical Biology*, *152*(3), pp.329-339.

Bozec, Y.M., Kulbicki, M., Laloë, F., Mou-Tham, G. and Gascuel, D., 2011. Factors affecting the detection distances of reef fish: implications for visual counts. *Marine Biology*, *158*(5), pp.969-981.

Brandt, S.B., Mason, D.M., Patrick, E.V., Argyle, R.L., Wells, L., Unger, P.A. and Stewart, D.J., 1991. Acoustic measures of the abundance and size of pelagic planktivores in Lake Michigan. *Canadian Journal of Fisheries and Aquatic Sciences*, *48*(5), pp.894-908.

Coll, C., de Morais, L.T., Laë, R., Lebourges-Dhaussy, A., Simier, M., Guillard, J., Josse, E., Ecoutin, J.M., Albaret, J.J., Raffray, J. and Kantoussan, J., 2007. Use and limits of three methods for assessing fish size spectra and fish abundance in two tropical man-made lakes. *Fisheries Research*, *83*(2), pp.306-318.

Daan, N., Gislason, H., G. Pope, J. and C. Rice, J., 2005. Changes in the North Sea fish community: evidence of indirect effects of fishing?. *ICES Journal of marine Science*, *62*(2), pp.177-188.

de Kerckhove, D.T., Shuter, B.J. and Milne, S., 2015. Acoustically derived fish size spectra within a lake and the statistical power to detect environmental change. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.565-574.

Draštík, V., Kubečka, J., Čech, M., Frouzová, J., Říha, M., Juza, T., Tušer, M., Jarolím, O., Prchalová, M., Peterka, J. and Vašek, M., 2009. Hydroacoustic estimates of fish stocks in temperate reservoirs: day or night surveys?. *Aquatic Living Resources*, *22*(1), pp.69-77.

Dulvy, N.K., Polunin, N.V., Mill, A.C. and Graham, N.A., 2004. Size structural change in lightly exploited coral reef fish communities: evidence for weak indirect effects. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(3), pp.466-475.

Edwards, A.M., Robinson, J.P., Plank, M.J., Baum, J.K. and Blanchard, J.L., 2017. Testing and recommending methods for fitting size spectra to data. *Methods in Ecology and Evolution*, *8*(1), pp.57-67.

Edgar, G.J. and Stuart-Smith, R.D., 2009. Ecological effects of marine protected areas on rocky reef communities—a continental-scale analysis. *Marine Ecology Progress Series*, *388*, pp.51-62.

Egerton, J.P., Johnson, A.F., Le Vay, L., McCoy, C.M., Semmens, B.X., Heppell, S.A. and Turner, J.R., 2017. Hydroacoustics for the discovery and quantification of Nassau grouper (*Epinephelus striatus*) spawning aggregations. *Coral Reefs*, *36*(2), pp.589-600.

Ehrenberg, J.E. and Torkelson, T.C., 1990. Application of dual-beam and split-beam target tracking in fisheries acoustics. *Representation*, *29*(106), pp.329-334.

Emmrich, M., Helland, I.P., Busch, S., Schiller, S. and Mehner, T., 2010. Hydroacoustic estimates of fish densities in comparison with stratified pelagic trawl sampling in two deep, coregonid-dominated lakes. *Fisheries Research*, *105*(3), pp.178-186.

Engelhard, G.H., Peck, M.A., Rindorf, A., C. Smout, S., van Deurs, M., Raab, K., Andersen, K.H., Garthe, S., Lauerburg, R.A., Scott, F. and Brunel, T., 2013. Forage fish, their fisheries, and their predators: who drives whom?. *ICES Journal of Marine Science*, *71*(1), pp.90-104.

Foote, K. G., Knudsen, H. P., Vestnes, G., MacLennan, D. N., and Simmonds. E. J. 1987. Calibration of acoustic instruments for fish density estimation: a practical guide. Cooperative Research Report. *ICES Journal of marine Science, 144*, 69 pp

Friedlander, A.M., Brown, E.K. and Monaco, M.E., 2007. Coupling ecology and GIS to evaluate efficacy of marine protected areas in Hawaii. *Ecological Applications*, *17*(3), pp.715-730.

Froese, R., 2006. Cube law, condition factor and weight–length relationships: history, meta-analysis and recommendations. *Journal of applied ichthyology*, *22*(4), pp.241-253.

Froese R, Pauly D. 2016. FishBase. World Wide Web electronic publication. www.fishbase.org, version (06/2014).

Gislason, H., and Rice, J. 1998. Modelling the response of size and diversity spectra of fish assemblages to changes in exploitation. *ICES Journal of Marine Science*, *55*: 362–370.

Gómez-Canchong, P., Quiñones, R.A. and Brose, U., 2013. Robustness of size–structure across ecological networks in pelagic systems. *Theoretical Ecology*, *6*(1), pp.45-56.

Graham, N.A., Wilson, S.K., Jennings, S., Polunin, N.V., Robinson, J.A.N., Bijoux, J.P. and Daw, T.M., 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. *Conservation Biology*, *21*(5), pp.1291-1300.

Jacobsen, N.S., Gislason, H. and Andersen, K.H., 2014. The consequences of balanced harvesting of fish communities. *Proceedings of the Royal Society of London B: Biological Sciences*, *281*(1775), p.20132701.

Jennings, S. 2005. Size-based analyses of aquatic food webs. Aquatic food webs: an ecosystem approach. Oxford University Press, Oxford, pp.86-97.

Jennings, S. and Dulvy, N. K. 2005. Reference points and reference directions for size-based indicators of community structure. *ICES Journal of Marine Science 62*: 397–404.

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

Jennings, S. and Mackinson, S., 2003. Abundance–body mass relationships in size-structured food webs. *Ecology Letters*, 6(11), pp.971-974.

Jiang, Y.Z., Cheng, J.H. and Li, S.F., 2009. Temporal changes in the fish community resulting from a summer fishing moratorium in the northern East China Sea. *Marine Ecology Progress Series*, *387*, pp.265-273.

Jones, N., 2011. Little Mexican reserve boasts big recovery. *Nature*. Available at: http://www.nature.com/doifinder/10.1038/news.2011.479

Kracker, L., 2007. Hydroacoustic surveys: A non-destructive approach to monitoring fish distributions at National Marine Sanctuaries. NOAA Technical Memorandum NOS NCCOS 66

Law, R., Plank, M.J. and Kolding, J., 2012. On balanced exploitation of marine ecosystems: results from dynamic size spectra. *ICES Journal of Marine Science*, *69*(4), pp.602-614.

Love, R.H., 1971. Measurements of fish target strength: a review. *Fisheries Bulletin*, *69*(4), pp.703-715.

Mackinson, S., Freeman, S., Flatt, R. and Meadows, B., 2004. Improved acoustic surveys that save time and money: integrating fisheries and ground-discrimination acoustic technologies. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.129-140.

McDonald, J.H. 2014. Handbook of Biological Statistics (3rd ed.). Sparky House Publishing, Baltimore, Maryland. 213-219

McClanahan TR, Graham NAJ. 2005. Recovery trajectories of coral reef fish assemblages within Kenyan marine protected areas. *Marine Ecology Progress Series 294*: 241–248

Mourier, J., Maynard, J., Parravicini, V., Ballesta, L., Clua, E., Domeier, M.L. and Planes, S., 2016. Extreme inverted trophic pyramid of reef sharks supported by spawning groupers. *Current Biology*, *26*(15), pp.2011-2016.

Munyandorero, J., 2006. Size, diversity, dominance and trophic structures of the fish community in a tropical lake: a case study for the Sinazongwe area, Lake Kariba, Zambia. *Aquatic Living Resources*, *19*(2), pp.123-136.

Murawski, S.A., and Idoine, J.S. 1992. Multispecies size composition: a conservative property of exploited fishery systems? *Journal of Northwest Atlantic Fisheries Science*, *14*:79-85

Ona E and Mitson RB. 1996. Acoustic sampling and signal processing near the seabed: the deadzone revisited. *ICES Journal of Marine Science*; *53*, pp. 677–690

Parker-Stetter, S.L. Rudstam, L.G., Sullivan, P.J., Warner, D.M. 2009. Standard operating procedures for fisheries acoustic surveys in the Great Lakes. Great Lakes Fisheries Commission Special Publication 09-01

Pollom, R.A. and Rose, G.A., 2015. Size-based hydroacoustic measures of within-season fish abundance in a boreal freshwater ecosystem. *PloS One*, *10*(4), p.e0124799.

Pope, J.G. and Knights, B.J., 1982. Comparison of length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *Multispecies approaches to fisheries management advice*, *59*, pp.116-118.

Reid, D.G. 2000. ICES Cooperative Research Report No. 238. 2000. Report on Echo Trace Classification. David G. Reid (ed). Copenhagen, Denmark.

Rice, J.C. 2000. Evaluating fishery impacts using metrics of community structure. *ICES Journal of Marine Science*, 57, 682–688.

Rice, J. and Gislason, H., 1996. Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. *ICES Journal of Marine Science*, *53*(6), pp.1214-1225.

Riegl, B.M., Halfar, J., Purkis, S.J. and Godinez-Orta, L., 2007. Sedimentary facies of the eastern Pacific's northernmost reef-like setting (Cabo Pulmo, Mexico). *Marine Geology*, *236*(1), pp.61-77.

Robinson, J.P., Williams, I.D., Edwards, A.M., McPherson, J., Yeager, L., Vigliola, L., Brainard, R.E. and Baum, J.K., 2017. Fishing degrades size structure of coral reef fish communities. *Global Change Biology*, *23*(3), pp.1009-1022.

Rochet, M.J., Benoît, E., Blanchard, F., Boucher, J., Poulard, J.C., Souplet, A. and Vérin, Y., 1999. La pente de la structure en taille multispécifique: un indicateur de l'effet de la pêche sur les peuplements de poissons exploités. *Rapport final. Programme interdisciplinaire de recherche" Environnement, vie et sociétés". Méthodologies et pertinence d'indicateurs d'interactions populations humaines-milieux.*

Rogers, A., Blanchard, J.L. and Mumby, P.J., 2014. Vulnerability of coral reef fisheries to a loss of structural complexity. *Current Biology*, *24*(9), pp.1000-1005.

Rudstam, L.G., Parker-Stetter, S.L., Sullivan, P.J. and Warner, D.M., 2009. Towards a standard operating procedure for fishery acoustic surveys in the Laurentian Great Lakes, North America. *ICES Journal of Marine Science*, *66*(6), pp.1391-1397.

Sandin, S.A., Smith, J.E., DeMartini, E.E., Dinsdale, E.A., Donner, S.D., Friedlander, A.M., Konotchick, T., Malay, M., Maragos, J.E., Obura, D. and Pantos, O., 2008. Baselines and degradation of coral reefs in the Northern Line Islands. *PloS One*, *3*(2), p.e1548.

Sawada, K., Furusawa. M., Williamson. N.J., 1993. Conditions for the precise measurement of fish target strength in situ. *Journal of the Marine Acoustical Society. 20*; 15–21.

Shin, Y.J. and Cury, P., 2004. Using an individual-based model of fish assemblages to study the response of size spectra to changes in fishing. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(3), pp.414-431.

Shin, Y.J., Rochet, M.J., Jennings, S., Field, J.G. and Gislason, H., 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. *ICES Journal of Marine Science 62*(3), pp.384-396.

Smyth, D., Al-Maslamani, I., Chatting, M. and Giraldes, B., 2016. Benthic surveys of the historic pearl oyster beds of Qatar reveal a dramatic ecological change. *Marine Pollution Bulletin*, *113*(1), pp.147-155.

Sprules, W.G. and Barth, L.E., 2015. Surfing the biomass size spectrum: some remarks on history, theory, and application. *Canadian Journal of Fisheries and Aquatic Sciences*, 73(4), pp.477-495.

Sweeting, C.J., Badalamenti, F., D'Anna, G., Pipitone, C. and Polunin, N.V.C., 2009. Steeper biomass spectra of demersal fish communities after trawler exclusion in Sicily. *ICES Journal of Marine Science*, *66*(1), pp.195-202.

Trebilco, R., Baum, J.K., Salomon, A.K. and Dulvy, N.K., 2013. Ecosystem ecology: size-based constraints on the pyramids of life. *Trends in Ecology and Evolution*, *28*(7), pp.423-431.

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C., 2011. Underwater acoustics for ecosystem-based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, *442*, pp.285-301.

Ward-Paige, C., Flemming, J.M. and Lotze, H.K., 2010. Overestimating fish counts by non-instantaneous visual censuses: consequences for population and community descriptions. *PLoS One*, *5*(7), p.e11722.

Wheeland, L.J. and Rose, G.A., 2015. Acoustic measures of lake community size spectra. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.557-564.

Wilson, S.K., Fisher, R., Pratchett, M.S., Graham, N.A.J., Dulvy, N.K., Turner, R.A., Cakacaka, A. and Polunin, N.V.C., 2010. Habitat degradation and fishing effects on the size structure of coral reef fish communities. *Ecological Applications*, *20*(2), pp.442-451.

Winfield, I.J., Emmrich, M., Guillard, J., Mehner, T. and Rustadbakken, A., 2011. WISER Deliverable 3.4-3: Guidelines for standardisation of hydroacoustic methods. *WISER Project Newsletter*, *2*, pp.4-4.

Yemane, D., Field, J.G. and Leslie, R.W., 2010. Spatio-temporal patterns in the diversity of demersal fish communities off the south coast of South Africa. *Marine Biology*, *157*(2), pp.269-281.

Yule, D.L., Adams, J.V., Hrabik, T.R., Vinson, M.R., Woiak, Z. and Ahrenstorff, T.D., 2013. Use of classification trees to apportion single echo detections to species: application to the pelagic fish community of Lake Superior. *Fisheries Research*, *140*, pp.123-132.

Yurista, P.M., Yule, D.L., Balge, M., VanAlstine, J.D., Thompson, J.A., Gamble, A.E., Hrabik, T.R., Kelly, J.R., Stockwell, J.D. and Vinson, M.R., 2014. A new look at the Lake Superior biomass size spectrum. *Canadian Journal of Fisheries and Aquatic Sciences*, *71*(9), pp.1324-1333.

Zgliczynski, B.J. and Sandin, S.A., 2017. Size-structural shifts reveal intensity of exploitation in coral reef fisheries. *Ecological Indicators*, *73*, pp.411-421.

Zwanenburg, K.C.T., 2000. The effects of fishing on demersal fish communities of the Scotian Shelf. *ICES Journal of Marine Science*, *57*(3), pp.503-509.

7 Chapter 7. Synthesis

7.1 Efficacy of Hydroacoustics

Hydroacoustics offers many advantages for surveying fish communities over traditional methods such as UVC and fishing/trawling. These advantages include that coverage possible per unit time is greatly increased, data is recorded instantly, there is minimum observer error, and almost the whole water column can be surveyed (Trenkel et al. 2011; Yurista et al. 2014). Further there are no issues due to water clarity, diver depth limits and also importantly hydroacoustic methods are non-destructive (Murphy and Jenkins, 2010). They do however, have their own suite of biases and limitations (Zenone et al. 2017). The greatest limitation being that without highly rigorous groundtruthing, fish densities, lengths and subsequent biomass estimates are likely to contain sources of error. This is resultant from taking mean values of TS which varies between fish species and sizes, Sv is divided by. Further, it is well known that TS can also vary due to other factors such as fish behaviour and tilt angle and should therefore be considered as a stochastic variable. As such, all fish densities, lengths and biomasses described in this thesis should be taken as estimates. However, throughout this thesis it is argued that once these sources of error are acknowledged, these estimates can still provide managers with highly useful information with which to make management decisions (Boswell, 2007). Groundtruthing methods are most commonly extractive (e.g. trawling) and this is often not possible in an MPA or sensitive habitat situation. In Chapter 3, detailed groundtruthing was enabled via use of both the towed camera system and through length based data provided by SCUBA divers using laser callipers. This enabled a site- and species-specific TS-length formula to be derived which may be of much use for future hydroacoustic monitoring surveys.

There are two options to derive *in situ* Target Strength (TS) in the fish acoustics, either to use 'Single Echoes Detected' (SED) or to use 'Tracked fish' which is a consecutive string of SEDs defined as coming from one fish (Balk and Lindem, 2006). Tracked fish should provide better estimates of fish TS as a mean value can be determined and also can be used to determine tilt angle (Ehrenberg and Torkelson, 1996). SEDs are however useful as on occasion there are too few confirmed fish tracks available for a reliable mean *in situ* TS per analysis cell to be calculated (Balk and Lindem, 2006). As both methods use the same underlying data to find a mean TS, the results should however be similar. These different processing methods are examined in Chapters 3 (Cayman) and 6 (Size Spectra). In Chapter 3, differences between using SEDs and Tracked fish are examined in addition into the 75th percentile of Tracked fish.

Here it was found there was no difference between using SED and Tracked fish for the different sites, although the 75th percentile values were higher as would be expected. In Chapter 6, the differences between using SED and Tracked fish to derive size spectra indices are examined and there was a significant relationship between the resultant slopes and midpoints from using the two datasets. This shows that in the chapters where SEDs alone were used (Chapters 3 and 4), the results are likely to be similar to if Tracked fish were used. In providing estimates of fish density, lengths and biomass and additionally habitat, it is possible that alternative software options (e.g. Echoview (www.echoview.com)) may have yielded slightly different results. Unfortunately however, this was not available for data analyses, but would have been interesting to examine. Both software options however use the same underlying data and equations, so the results in theory should be the same. One aspect in Echoview that would have been useful is the option to take depth out of the clustering process to derive habitat classes. This would have been interesting to examine if the different depths of the different habitat types in Chapter 5 had an overriding effect on the processed classes.

In terms of comparison between different survey methods, hydroacoustic abundance estimates were compared with estimates from UVC in Chapters 3 (Cayman) and 4 (Mexico). Such comparisons are rare in the literature, although Taylor et al. (2006) did attempt this on the same Fish Spawning Aggregation (FSA) as in Chapter 3. Further, Zenone et al. (2017) recently conducted such a comparison and found that estimates between methods were correlated, but diver estimates were an order of magnitude higher. Taylor et al. (2006) conducting similar research to Chapter 3 in this thesis, reported similar acoustic density estimates to diver estimates over their entire survey region, although total abundances differed likely due to differences in survey coverage and the patchy distribution of the fish. In Chapter 3, no statistical difference was detected in abundance estimates using the two methods, although it should be noted that the number of events to compare the two methods was limited. In Chapter 4, UVC estimates were significantly higher but biomass estimates were not significantly different. This was due to the UVC surveys recording high numbers of small fish in comparison to hydroacoustics and the potential reasons for this were discussed.

7.2   MPA effects

Despite the recognised potential of using hydroacoustics to examine the efficacy of Marine Protected Areas (MPA) on fish populations (Kracker, 2007; Polunin et al. 2009), no references can be found in the literature of any research doing so. Determining the effects of MPAs on

fish communities through hydroacoustics were investigated primarily in Chapter 4 (Mexico). These data were then also examined through a size spectra approach in Chapter 6. In Chapter 4, fish abundance estimates were four times higher, and significantly so, in the Cabo Pulmo National Park (CPNP) in comparison to fished control areas of similar habitat. This is a finding that is in line with previous studies reporting the high fish biomass inside the CPNP (Aburto-Oropeza et al. 2011) and also with most meta-analyses on MPA effects (Halpern, 2003; Lester et al. 2009; Sciberras et al. 2013). Calculated biomass values were also significantly higher within the CPNP as was the mean size of fish, something that has been noted by other authors (e.g. Edgar and Stuart-Smith, 2009). Hydroacoustic surveys, such as conducted here, may be a valuable tool for managers endeavouring to monitor fish populations within MPAs, something that is known to be a difficult task (Hill et al. 2014). Repeating hydroacoustics on a temporal basis could help to reveal both the variability in hydroacoustic estimates and also how the fish population varies in a temporal aspect such as between seasons or years. Hydroacoustics could also be used to examine spillover effects from MPAs and potentially the differences between the protection afforded by the MPA on pelagic vs demersal populations. Throughout this thesis, however it is recommended that there would be considerable merit in nesting UVC surveys within a hydroacoustic survey campaign, to provide higher resolution species-specific information in conjunction with the broader scale estimates of fish density, biomass and size (see also Murphy and Jenkins, 2010).

The protection that MPAs afford to the FSAs that they are designed to safeguard was however also examined in a spatial context in Chapter 3. Here, it was found that at one of the FSA sites, the aggregation had elements outside of the protected zone and at another the FSA was very close to the boundary of the MPA. If fishing occurs at the boundary of these protected areas, as is known to occur following closures to fishing (Kellner et al. 2007), then it is possible that these FSAs, which may be recovering, could still be vulnerable. It was therefore recommended that the Cayman Islands Government consider slightly increasing the size (~100m) of the MPA. Further, although the Qatar sites in Chapter 5 are not currently afforded MPA status, the surveys described were also done so in order to identify potential suitable areas to place MPAs in the future. This dataset will provide highly useful in the 'Before After Control Impact' (BACI) approach (Bernstein and Zalinski, 1983) to rigorously assess any effects that any new management regime may have on fish populations.

7.3 Habitat effects

Increased habitat complexity has been shown to have a positive effect on fish abundance and biomass (Luckhurst and Luckhurst, 1978; Grigg, 1994; Komyakova et al. 2013; Graham and Nash, 2013). Using hydroacoustics to examine the effects of habitat type was examined primarily in both Chapters 4 (Mexico) and 5 (Qatar). The data from these chapters was then further interrogated through size spectra analyses (Chapter 6) described below. In Chapter 4, in addition to the MPA effect described above, habitat effects were also examined. Through this we found that more complex habitats had a greater biomasses and mean fish sizes. Between the control sites, the sandy control had a higher abundance of fish than the rocky control due to increased abundances in the smallest size classes, showing how small fish do not contribute greatly to total biomass values (Wanzenböck et al. 2003). Within the CPNP, the reefs had an order of magnitude greater fish abundance and biomass than the other habitat types present in the park. This demonstrates the high importance of these reefs for the National Park and the area in general. Examining the effects of habitat through hydroacoustic methods was taken further in Chapter 5 (Qatar), where the sampling strategy was based upon habitat type in order to examine this specifically. Similar to Chapter 4, greater abundances of fish, and an acoustic proxy for biomass (sv) found in sites with more complex habitats of the Gulf. Here no attempt was made to convert this biomass proxy into t/ha values as comparison with UVC estimates were not within the scope of the surveys. The more rugose sites here were found to have a larger average fish sizes. In agreement with other authors (e.g. Carpenter et al. 1981; Bell and Galzin, 1984; Graham and Nash, 2013), the site with the highest abundance of live coral also had the greatest abundance of fish. Different processing methods and software were examined to investigate if the strength of the seabed echo were better capable of describing fish distribution than depth. This was the case with Biosonics' Visual Habitat software, and although depth dependency may still be an issue, the ability to extract bottom type from the same survey data as fish shows potential and has important time-saving implications (Freeman et al. 2004; Mackinson et al. 2004).

By using the Tracked fish option described above, fish heights in the water column over the different habitat types could be examined. Examination of fish heights in the water column has been conducted in examining diel migrations of fish populations previously (e.g. Bohl, 1980), but no reference on using such data to investigate fish–seabed affiliation can be found in the literature. Using this approach, the mean height of fish in the water column was shown to be

closer to the seabed at the more complex 'shallow' category sites, potentially demonstrating stronger links with this habitat in comparison to the softer sediment less complex areas. Further, the analyses showed that larger fish had stronger links to habitat than small fish at all sites but strongest over most complex habitats. This is in contrast to most other authors who find smaller fish over more complex habitats (e.g. Graham et al. 2007; Wilson et al. 2010), but in agreement with some who have noted increasing fish size with increased complexity (Friedlander and Parrish, 1998; Boswell et al. 2007) and is likely to be worthy of additional research.

7.4 Size spectra

Size-based analyses, where body size is the principal descriptor in a trophic food web rather than species identity is a well-established analysis conducted on fish communities (Jennings, 2005; Bianchi et al. 2000; Rice, 2000). The data from Chapters 3, 4 and 5 are re-examined through a size spectra approach in Chapter 6. Using hydroacoustic methods for size-based approaches has the major advantage (in addition to those discussed above) of not being size selective (Shin et al. 2005). In traditional size spectra approaches through fishing, small fish are often not retained by the sampling gear and as such the resultant size spectra does not represent the entire community (Jennings and Dulvy, 2005; Yurista et al. 2014). UVC methods can also introduce error in the counts of small or large fish size classes (McClanahan and Graham, 2005; Bozec et al. 2011, Ward-Paige et al. 2010). The advantage of hydroacoustic data has therefore led to it being used in freshwater environments (Brandt et al. 1991; Yule et al. 2013; Wheeland and Rose, 2015). The analyses conducted in Chapter 6 are possibly the first time that size spectra derived from hydroacoustics has been conducted with marine fish communities. Through this it was found that the spectra tended to fit a quadratic relationship better than a linear one, which has been noted as an open question in the literature (Shin et al. 2005). This is an important finding in interpreting acoustically-derived fish spectra data and may help further in the understanding of size spectra in general. Further, this curvilinear nature of the spectra was more pronounced over areas of more rugose habitat, a finding that is in agreement with other recent research (Rogers et al. 2014). The heights of the spectra were also greatest over more complex habitats, further demonstrating the increased abundance of fish with increased habitat complexity (e.g. Wilson et al. 2010; Alvarez-Filip et al. 2011). When linear models were fitted through the Chapter 4 data (Mexico), slopes were steepest in control areas open to fishing, following previous studies (e.g. Blanchard et al. 2009; Robinson et al. 2016). This may be due to the removal of larger fish through size-selective fishing practices,

and the corresponding 'prey release' of smaller fishes (Sweeting et al. 2009). With Chapter 5 data (Qatar), slopes were steepest over more complex habitats which may again indicate fishing targeting these sites and removing larger fish. Contrarily, comparing the Chapter 4 controls, a steeper slope was seen over the less complex habitat site, which may however be due to unknown and differing levels of fishing pressure. Examination of size spectra from hydroacoustic data between sites with similar habitat and known levels of fishing intensity and also across temporal scales would be worthy of future research. Hydroacoustics coupled with a size spectra approach could be used as a valuable tool for which to assess and monitor changes in fish populations in a fast and non-destructive manner.

7.5 Concluding remarks

Whilst hydroacoustic fish surveys are routinely conducted for broad scale fish surveys and the effects of MPAs are well known, there has been extremely limited use of hydroacoustics to examine MPA effects. The overall aim of this study was to examine how hydroacoustics can be used to assess fish within existing and candidate MPAs. This was achieved at study sites in the Cayman Islands, Mexico and Qatar. In the Cayman Islands fish spawning aggregations (FSAs) were successfully quantified and examined in relation to the MPAs designed to protect them. In Mexico hydroacoustics were utilised to demonstrate high values of fish density and biomass within the park in comparison to suitable control areas. In Qatar, Government efforts to move towards Ecosystem Based Fisheries Management (EBFM) resulted in the surveys of areas which may achieve future protected status, providing data that may serve as a baseline from which any changes due to management can be assessed. The research described in this thesis includes many novel aspects. The first hydroacoustic assessment of an MPA is detailed in Chapter 4 (Mexico), the first use of hydroacoustic derived marine fish size spectra in Chapter 6 and novel methods to examine fish distribution in relation to habitat in Chapter 5 (Qatar). The lack of research in this area may be due to difficulties in groundtruthing acoustics data to a species level, but it is shown that hydroacoustics can still provide important ecological answers even when only limited groundtruthing is available. It is hoped that these additions to the scientific record will be of use to managers and others working and researching in EBFM in efforts to conserve the last remaining significant source of wild food, fish (Jacquet et al. 2010).

7.6   References

Aburto-Oropeza, O., Erisman, B., Galland, G.R., Mascareñas-Osorio, I., Sala, E. and Ezcurra, E., 2011. Large recovery of fish biomass in a no-take marine reserve. *PLoS One*, *6*(8), p.e23601.

Alvarez-Filip, L., Reyes-Bonilla, H. and Calderon-Aguilera, L.E., 2006. Community structure of fishes in Cabo Pulmo reef, Gulf of California. *Marine Ecology*, *27*(3), pp.253-262.

Balk, H. and Lindem, T. 2006. Sonar 4, Sonar 5, Sonar 6 e Post-processing Systems. Operator Manual. Oslo: Lindem Data Acquisition.

Bell, J.D. and Galzin, R., 1984. Influence of live coral cover on coral-reef fish communities. *Marine Ecology Progress Series*, pp.265-274.

Bernstein, B.B. and Zalinski, J., 1983. An optimum sampling design and power tests for environmental biologists. *Journal of Environmental Management*, *16*(1), pp.35-43.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Jin, X., Koranteng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F. and Zwanenburg, K., 2000. Impact of fishing on size composition and diversity of demersal fish communities. *ICES Journal of Marine Science*, *57*(3), pp.558-571.

Blanchard, J.L., Jennings, S., Law, R., Castle, M.D., McCloghrie, P., Rochet, M.J. and Benoît, E., 2009. How does abundance scale with body size in coupled size-structured food webs?. *Journal of Animal Ecology*, *78*(1), pp.270-280.

Boswell, K.M., Wilson, M.P. and Wilson, C.A., 2007. Hydroacoustics as a tool for assessing fish biomass and size distribution associated with discrete shallow water estuarine habitats in Louisiana. *Estuaries and Coasts*, *30*(4), pp.607-617.

Bohl, E. 1980. Diel pattern of pelagic distribution and feeding in planktivorous fish. *Oecologia 44*: 368-375

Bozec, Y.M., Kulbicki, M., Laloë, F., Mou-Tham, G. and Gascuel, D., 2011. Factors affecting the detection distances of reef fish: implications for visual counts. *Marine Biology*, *158*(5), pp.969-981.

Brandt, S.B., Mason, D.M., Patrick, E.V., Argyle, R.L., Wells, L., Unger, P.A. and Stewart, D.J., 1991. Acoustic measures of the abundance and size of pelagic planktivores in Lake Michigan. *Canadian Journal of Fisheries and Aquatic Sciences*, *48*(5), pp.894-908.

Carpenter, K.E., Miclat, R.I., Albaladejo, V.D. and Corpuz, V.T., 1981, May. The influence of substrate structure on the local abundance and diversity of Philippine reef fishes. In *Proceedings of the 4th International Coral Reef Symposium* (Vol. 2, pp. 497-502).

Edgar, G.J. and Stuart-Smith, R.D., 2009. Ecological effects of marine protected areas on rocky reef communities—a continental-scale analysis. *Marine Ecology Progress Series*, *388*, pp.51-62.

Ehrenberg, J.E. and Torkelson. T.C. 1996. Application of dual-beam and split-beam target tracking in fisheries acoustics. *ICES Journal of Marine Science, 53* (1996), pp. 329–334

Freeman, S., Mackinson, S. and Flatt, R., 2004. Diel patterns in the habitat utilisation of sandeels revealed using integrated acoustic surveys. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.141-154.

Friedlander, A.M. and Parrish, J.D., 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *Journal of Experimental Marine Biology and Ecology*, *224*(1), pp.1-30.

Graham, N.A.J., Nash, K.I. 2013. The importance of structural complexity in coral reef ecosystems. *Coral Reefs 32*: 315−326

Graham, N.A., Wilson, S.K., Jennings, S., Polunin, N.V., Bijoux, J.P. and Robinson, J., 2006. Dynamic fragility of oceanic coral reef ecosystems. *Proceedings of the National Academy of Sciences*, *103*(22), pp.8425-8429.

Grigg, R.W., 1994. Effects of sewage discharge, fishing pressure and habitat complexity on coral ecosystems and reef fishes in Hawaii. *Marine Ecology Progress Series*, pp.25-34.

Halpern, B.S., 2003. The impact of marine reserves: do reserves work and does reserve size matter?. *Ecological Applications*, pp.S117-S137.

Hill, N.A., Barrett, N., Lawrence, E., Hulls, J., Dambacher, J.M., Nichol, S., Williams, A. and Hayes, K.R., 2014. Quantifying fish assemblages in large, offshore marine protected areas: an Australian case study. *PloS One*, *9*(10), p.e110831

Jacquet, J., Hocevar, J., Lai, S., Majluf, P., Pelletier, N., Pitcher, T., Sala, E., Sumaila, R. and Pauly, D., 2010. Conserving wild fish in a sea of market-based efforts. *Oryx*, *44*(1), pp.45-56.

Jennings, S., 2005. Size-based analyses of aquatic food webs. Aquatic food webs: an ecosystem approach. Belgrano, A., Scharler, UM, Dunne, J. and Ulanowicz, RE. Oxford University Press, Oxford, pp.86-97.

Jennings, S. and Dulvy, N.K., 2005. Reference points and reference directions for size-based indicators of community structure. *ICES Journal of Marine Science*, *62*(3), pp.397-404.

Kellner, J.B., Tetreault, I., Gaines, S.D. and Nisbet, R.M., 2007. Fishing the line near marine reserves in single and multispecies fisheries. *Ecological Applications*, *17*(4), pp.1039-1054.

Komyakova, V., Munday, P.L. and Jones, G.P., 2013. Relative importance of coral cover, habitat complexity and diversity in determining the structure of reef fish communities. *PLoS One*, *8*(12), p.e83178.

Kracker, L., 2007. Hydroacoustic surveys: A non-destructive approach to monitoring fish distributions at National Marine Sanctuaries. NOAA Technical Memorandum NOS NCCOS 66

Lester, S.E., Halpern, B.S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B.I., Gaines, S.D., Airamé, S. and Warner, R.R., 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, *384*, pp.33-46.

Luckhurst, B.E. and Luckhurst, K., 1978. Analysis of the influence of substrate variables on coral reef fish communities. *Marine Biology*, *49*(4), pp.317-323.

Mackinson, S., Freeman, S., Flatt, R. and Meadows, B., 2004. Improved acoustic surveys that save time and money: integrating fisheries and ground-discrimination acoustic technologies. *Journal of Experimental Marine Biology and Ecology*, *305*(2), pp.129-140.

McClanahan, T.R, Graham, N.A.J. 2005. Recovery trajectories of coral reef fish assemblages within Kenyan marine protected areas. *Marine Ecology Progress Series 294*: 241–248

Murphy, H.M. and Jenkins, G.P., 2010. Observational methods used in marine spatial monitoring of fishes and associated habitats: a review. *Marine and Freshwater Research*, *61*(2), pp.236-252

Polunin, N., Bloomfield, H., Sweeting, C. and McCandless, D., 2009. Developing indicators of MPA effectiveness: finfish abundance and diversity in a Yorkshire prohibited trawl area. *Report to MFA, London*.

Rice, J.C., 2000. Evaluating fishery impacts using metrics of community structure. *ICES Journal of Marine Science*, *57*(3), pp.682-688.

Robinson, J.P., Williams, I.D., Edwards, A.M., McPherson, J., Yeager, L., Vigliola, L., Brainard, R.E. and Baum, J.K., 2017. Fishing degrades size structure of coral reef fish communities. *Global Change Biology*, *23*(3), pp.1009-1022.

Rogers, A., Blanchard, J.L. and Mumby, P.J., 2014. Vulnerability of coral reef fisheries to a loss of structural complexity. *Current Biology*, *24*(9), pp.1000-1005.

Sciberras, M., Jenkins, S.R., Mant, R., Kaiser, M.J., Hawkins, S.J. and Pullin, A.S., 2015. Evaluating the relative conservation value of fully and partially protected marine areas. *Fish and Fisheries*, *16*(1), pp.58-77.

Shin, Y.J., Rochet, M.J., Jennings, S., Field, J.G. and Gislason, H., 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. *ICES Journal of Marine Science*, *62*(3), pp.384-396.

Sweeting, C.J., Badalamenti, F., D'Anna, G., Pipitone, C. and Polunin, N.V.C., 2009. Steeper biomass spectra of demersal fish communities after trawler exclusion in Sicily. *ICES Journal of Marine Science*, *66*(1), pp.195-202.

Taylor, J.C., Eggleston, D.B. and Rand, P.S. 2006. Nassau grouper (*Epinephelus striatus*) spawning aggregations: hydroacoustic surveys and geostatistical analysis. In: Emerging technologies for reef fisheries research and management. Seattle, WA, NOAA, pp. 18-25. (NOAA Professional Paper NMFS, 5)

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C., 2011. Underwater acoustics for ecosystem-based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, *442*, pp.285-301.

Wanzenböck, J., Mehner, T., Schulz, M., Gassner, H. and Winfield, I.J., 2003. Quality assurance of hydroacoustic surveys: the repeatability of fish-abundance and biomass estimates in lakes within and between hydroacoustic systems. *ICES Journal of Marine Science*, *60*(3), pp.486-492.

Ward-Paige, C., Flemming, J.M. and Lotze, H.K., 2010. Overestimating fish counts by non-instantaneous visual censuses: consequences for population and community descriptions. *PLoS One*, *5*(7), p.e11722.

Wheeland, L.J. and Rose, G.A., 2015. Acoustic measures of lake community size spectra. *Canadian Journal of Fisheries and Aquatic Sciences*, *73*(4), pp.557-564.

Wilson, S.K., Fisher, R., Pratchett, M.S., Graham, N.A.J., Dulvy, N.K., Turner, R.A., Cakacaka, A. and Polunin, N.V.C., 2010. Habitat degradation and fishing effects on the size structure of coral reef fish communities. *Ecological Applications*, *20*(2), pp.442-451.

Yule, D.L., Adams, J.V., Hrabik, T.R., Vinson, M.R., Woiak, Z. and Ahrenstorff, T.D., 2013. Use of classification trees to apportion single echo detections to species: application to the pelagic fish community of Lake Superior. *Fisheries research*, *140*, pp.123-132.

Yurista, P.M., Yule, D.L., Balge, M., VanAlstine, J.D., Thompson, J.A., Gamble, A.E., Hrabik, T.R., Kelly, J.R., Stockwell, J.D. and Vinson, M.R., 2014. A new look at the Lake Superior biomass size spectrum. *Canadian Journal of Fisheries and Aquatic Sciences*, *71*(9), pp.1324-1333.

Zenone, A.M., Burkepile, D.E. and Boswell, K.M., 2017. A comparison of diver vs. acoustic methodologies for surveying fishes in a shallow water coral reef ecosystem. *Fisheries Research*, *189*, pp.62-66.