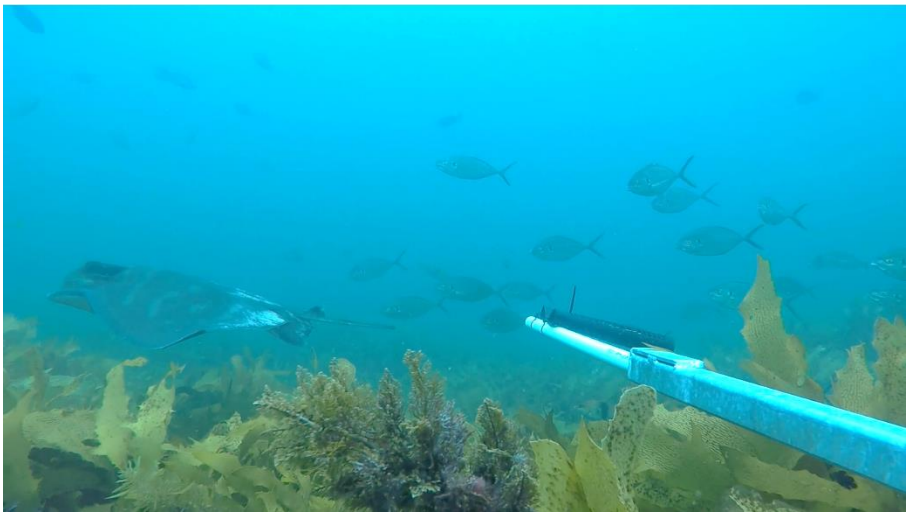
Quantifying Fish Assemblages in South Australian Marine Protected Areas



Nina Wootton

Supervisor: Bronwyn Gillanders

School of Biological Sciences

The University of Adelaide

Submitted in partial fulfilment of the requirements for the degree of

Bachelor of Science (Honours)

October 2017

THE UNIVERSITY of ADELAIDE

DECLARATION

I hereby declare that this work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference is made in the text.

Nina Wootton

October 2017

TABLE OF CONTENTS

ABSTRACT

Networks of Marine Protected Areas (MPAs) and no-take sanctuary zones are increasingly being developed as a means of conserving biodiversity and protecting the ocean. This study examined the abundance, species richness and lengths of fish within a network of MPAs in South Australia's west coast embayments within the first three years of development. Fish assemblages in the embayments were also examined to evaluate how biological and environmental conditions influenced them. Baited Remote Underwater Video (BRUV) were used to quantify fish assemblages in a range of depths and habitats over two seasons during 2017. Total fish abundance was 76% higher inside sanctuary zones compared to outside in both Coffin Bay and Venus Bay, although not all species, seasons and sanctuary zones performed equally. The key differences occurred between fish assemblages in different seasons (January and June) and in different sanctuary zones. Ideally, MPAs should incorporate a range of different habitats over a wide depth range. This may require larger sanctuary zones, or researching locations of potential MPAs prior to their establishment, to ensure that the protected regions cover a variety of habitats and depths.

INTRODUCTION

Human actions, including overfishing, pollution and maritime industries, have steadily decreased the healthy functioning of marine and estuarine habitats (Lotze *et al.* 2006; Crain *et al.* 2009). Due to increased awareness and global concern for marine ecosystem health, there has been an escalation in the development of Marine Protected Areas (MPAs) in the last decade (Halpern *et al.* 2010). Of these protected areas, a small amount are no-take zones which prohibit the extraction of marine resources in entirety (Gaines *et al.* 2010). Various published studies have researched and evaluated the response of a range of marine organisms and ecosystems to protection (Lester *et al.* 2009). These include researching the type of species that respond best to protection (Mosquera *et al.* 2000; Barrett *et al.* 2007), their impact on areas surrounding the protected zone (Russ and Alcala 2011; Harrison *et al.* 2012b), their influence on invasive species (Burfeind *et al.* 2013), the ideal setup of protected reserves (Claudet *et al.* 2008; Edgar *et al.* 2014), and the amount of enforcement needed for changes of a significant level to occur (Byers and Noonburg 2007; Guidetti *et al.* 2008).

Marine Protected Areas are important biodiversity conservation tools which protect natural and cultural resources. Quantitative information on fish assemblages in marine protected areas is required to assess change through time or to compare no take areas to areas where fishing is permitted. Such information can be used to evaluate the effectiveness of MPAs and gain community support. Limited data exists analysing fish assemblages in MPAs in South Australian waters, yet such data are critical to evaluate their effectiveness.

Worldwide less than 4% of the ocean is protected, with Australian waters accounting for 65% of the total global MPAs by area (Jenkins and Van Houtan 2016). Numerous studies conclude that protected areas contain higher fish diversity and abundance, with greater average size and general higher biodiversity than areas which are unprotected (Kelaher *et al.* 2014; Soler *et al.* 2015). Furthermore, there is strong evidence that there are community-wide changes surrounding MPAs, with increases in marine biota, improved larval export and recruitment benefits (Hilborn *et al.* 2004; Guidetti 2007; Harrison *et al.* 2012a). However, studies also suggest that MPAs may need to be in place for some years before these benefits are observed (Claudet *et al.* 2008; Edgar and Barrett 2012; Kelaher *et al.* 2014).

Marine Protected Areas vary in shape, size and location, and there has been limited investigation of how such variables may affect conservation goals (Edgar and Barrett 1999). The identification of significant scales of natural variability of fish species provides an important baseline for ecologically relevant choices on MPA design and evaluation of their effectiveness (Charton *et al.* 2002). These data are not available for a large number of marine species, especially at small spatial scales. The majority of studies focus either on fish assemblages over larger ranges (Parsons *et al.* 2016), or on individual species and trophic groups at small spatial scales (Gillanders 1997). Large spatial fish assemblage patterns are still useful for the broad-scale zoning and planning of MPAs, however are not effective in providing the resolution needed for zoning design decisions on MPAs at a local scale (Malcolm *et al.* 2007). Therefore, patterns of fish assemblage at small spatial scales are useful in identifying locations that need protection and can provide data to help implement this.

Spatial-temporal variability of habitats, and both the physical and biological attributes within these habitats (e.g. temperature and depth), are important influencers on the distribution of fish species locally (Choat and Ayling 1987; Kingsford 1989; Holbrook *et al.* 1994). The fish assemblage will vary depending on the habitat type within the environment. The type of habitat, such as seagrass, reef, macro algae or sand, are important factors determining whether areas are inhabited and therefore affect abundance. Seagrasses and mangroves in particular are often associated with juvenile nurseries (Beck *et al.* 2001; Heck *et al.* 2003; Dorenbosch *et al.* 2004), so it would be expected that fish size and composition would change in these habitats. Further physical factors, such as depth and temperature, can cause variations in fish assemblages within these habitats (Rooker and Dennis 1991; Parsons *et al.* 2016; Fitzgerald *et al.* 2017). Knowledge of the relationships between fish assemblages and habitat variability within small scale biogeographical regions would provide a more solid outline for designing MPAs.

South Australia has extensive areas of MPAs, under both Commonwealth and State legislation (Barr and Possingham 2013; Kirkman and Shepherd 2015). The state-controlled areas are legislated under the *South Australian Marine Parks Act 2007* and cover areas extending to three nautical miles from the coast (Barr and Possingham 2013). These state MPAs were developed in 2009 as part of a network of 19 marine parks with the ultimate aim

of conserving the unique marine life throughout the eight bioregions of South Australia (Kirkman 2013; Kirkman and Shepherd 2015). Within the MPAs there are zones with different levels of protection; these include restricted access, sanctuary and habitat protection zones (Lynch 2006; Kirkman 2013). Sanctuary zones are considered areas of high conservation value where no fishing or other disruptive activities are permitted, however enforcement of these areas did not commence until October 2014 (Scholz *et al.* 2017). Under the current legislation, *Marine Parks Act 2007*, the monitoring, evaluation and reporting of MPAs is required every 10 years (Scholz *et al.* 2017).

This study aims to assess MPA effectiveness by using Baited Remote Underwater Video (BRUV) to collect baseline fish assemblage data in South Australian west coast embayments. It specifically investigates if there are any significant differences in fish abundance, species richness and size inside and outside the sanctuary zones within the MPAs. Furthermore, it investigates patterns of fish assemblages at a small spatial scale, for example metres to kilometres, and looks at assemblage variation among habitats, and at which biological and physical factors are potentially contributing to these patterns.

It can be hypothesised that with time, there will be significant differences in fish abundance, species richness and size inside and outside MPAs. However, South Australian zoning was only implemented in October 2014. Therefore, differences in abundance, richness and size may not necessarily be found, as a global meta-analysis suggested that MPAs needed to be implemented for 10 years to be effective (Edgar *et al.* 2014). Despite this, the project provides useful baseline information upon which to detect change in the future. BRUV data may contribute to determining the success of MPAs within areas of South Australia.

MATERIALS AND METHODS

STUDY SITES

Nineteen multiple use marine parks were established in South Australia by the South Australian Government in 2012. This includes the West Coast Bays Marine Park including Venus Bay (Figure 1) and the Thorny Passage Marine Park including Coffin Bay (Figure 2). Within these marine parks are a number of sanctuary zones where fishing of any type is prohibited.

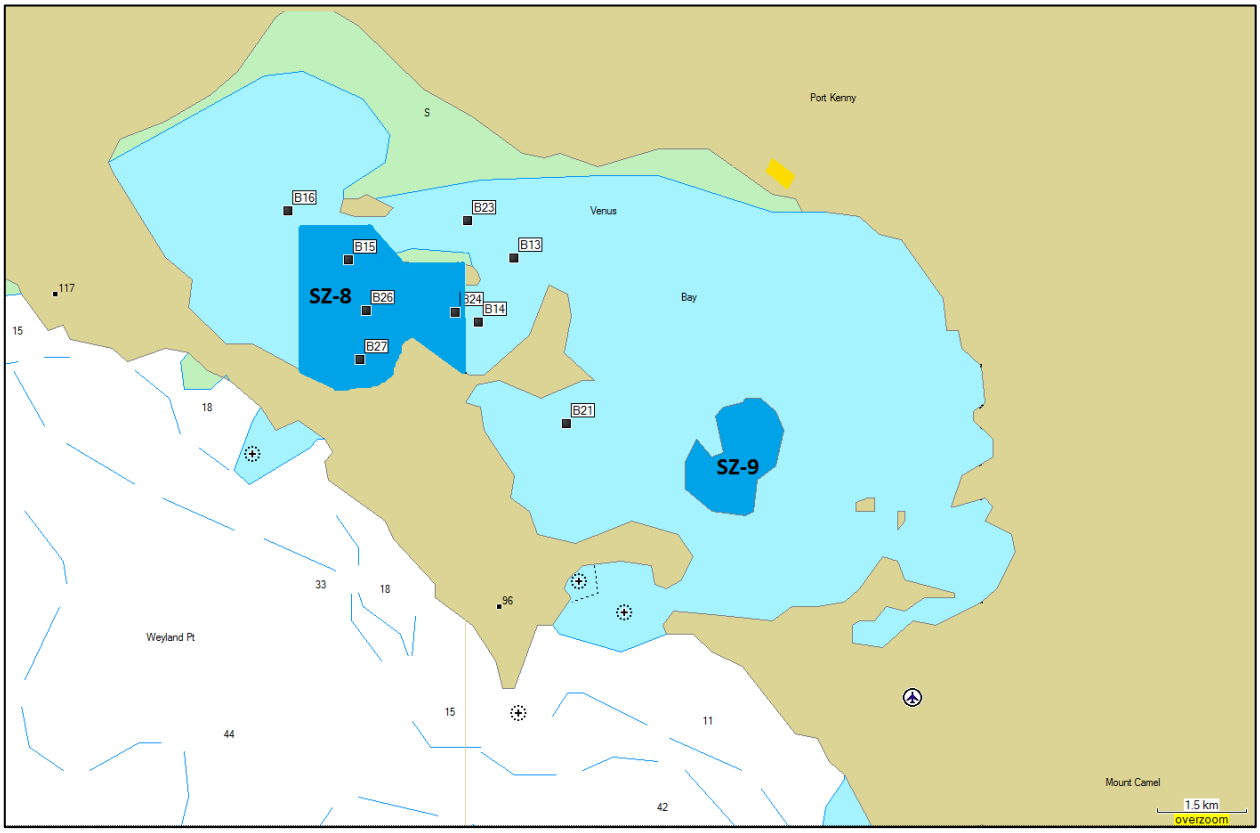


Figure 1. MPAs within the West Coast Bays Marine Park in South Australia, including the Venus Bay Sanctuary Zone 8 where sampling was completed. Note that Sanctuary Zone 9 does not include any subtidal habitat. Light blue area corresponds to MPAs while dark blue areas are sanctuary zones. Black GPS marks illustrate locations of BRUV deployments.

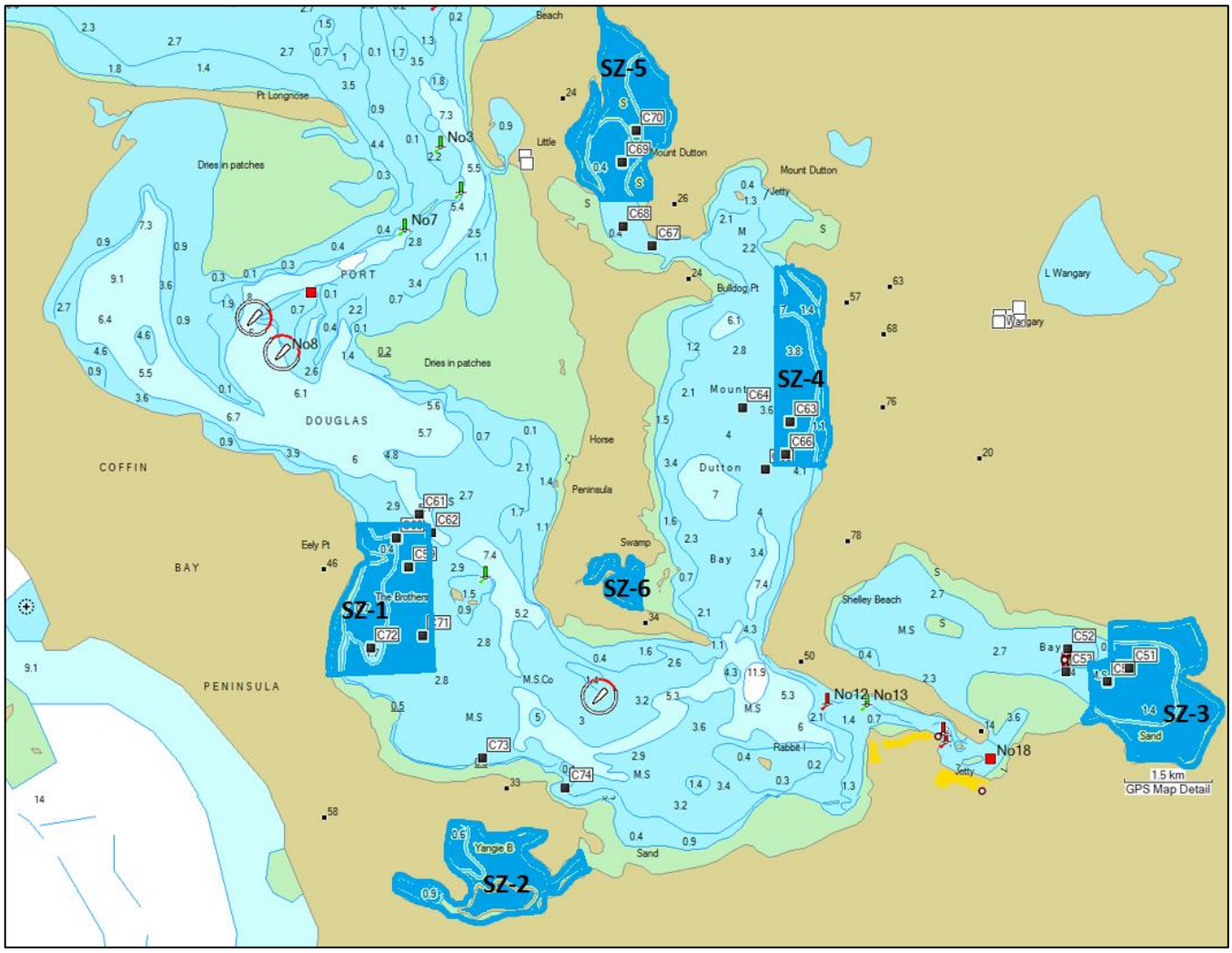


Figure 2. MPAs in the Thorny Passage Marine Park in South Australia, including Coffin Bay sanctuary zones where sampling was completed. This includes Sanctuary Zones 1, 3, 4 and 5. Sanctuary Zones 2 and 6 were excluded from the study due to being too small and too shallow for a boat to enter for BRUV sampling. Light blue areas correspond with MPAs while the dark blue areas are sanctuary zones. Black GPS marks illustrate locations of BRUV deployments.

In total, five sanctuary zones were sampled using BRUV cameras, including one in Venus Bay (Sanctuary Zone 8) and four in Coffin Bay (Sanctuary Zones 1, 3, 4 and 5). The number of BRUV deployments and replicates varied slightly between each sanctuary zone due to logistical and video recording issues. However, with the exception of Sanctuary Zone 4, there were at least two deployments inside each sanctuary zone and a matching replicate based on depth and habitat outside each sanctuary zone (Table 1). Sampling occurred in January and June 2017.

Table 1. Summary of information relating to sampling sites for January and June sampling. Sampling information includes: sanctuary zone, whether sampling was inside (I) or outside (O) the sanctuary zone and number of BRUV deployments.

	Sanctuary zone	Inside/Outside	Number of BRUV deployments
January			
Coffin Bay	1	I	2
Coffin Bay	1	O	1
Coffin Bay	3	I	2
Coffin Bay	3	O	2
Coffin Bay	4	I	0
Coffin Bay	4	O	2
Venus Bay	8	I	2
Venus Bay	8	O	3
June			
Coffin Bay	1	I	4
Coffin Bay	1	O	4
Coffin Bay	3	I	2
Coffin Bay	3	O	2
Coffin Bay	4	I	1
Coffin Bay	4	O	2
Coffin Bay	5	I	2
Coffin Bay	5	O	2
Venus Bay	8	I	4
Venus Bay	8	O	2

BAITED REMOTE UNDERWATER VIDEO

Stereo BRUVs were used to record fish assemblages, with two cameras mounted on each metal camera stand (Figure 3). The BRUV data were collected using GoPro Hero4 underwater digital video cameras. The cameras were placed in waterproof housings which were mounted on a metal frame. The housings sat above the substratum and faced horizontal and parallel to the ocean floor. A tube of polymerizing vinyl chloride (PVC) extended from the centre of the metal frame, with a mesh bait bag attached. The bait bag contained 500g of minced pilchards (*Sardinops sagax*) and sat 1200mm from the camera lens. The cameras faced forward and slightly inwards towards the bait bag. The units were deployed from a boat, with care taken to drop them into a similar depth and habitat inside and outside of the sanctuary zone. Following deployment, the boat was moved approximately 200 metres away

from the area before the next unit was dropped. This minimised the overlap of bait plumes, ensuring that double counts of fish did not occur. The cameras were left to film underwater for 60 minutes before being collected and deployed again. Four BRUV units were deployed concurrently, and following deployment the boat left the area to avoid any disturbances to fish activity. Procedures were similar to other BRUV research (Watson *et al.* 2005; Harvey *et al.* 2007; Watson *et al.* 2010), so that the data were comparable to other studies. Temperature was recorded using HOBO temperature loggers, which were attached to each BRUV unit.

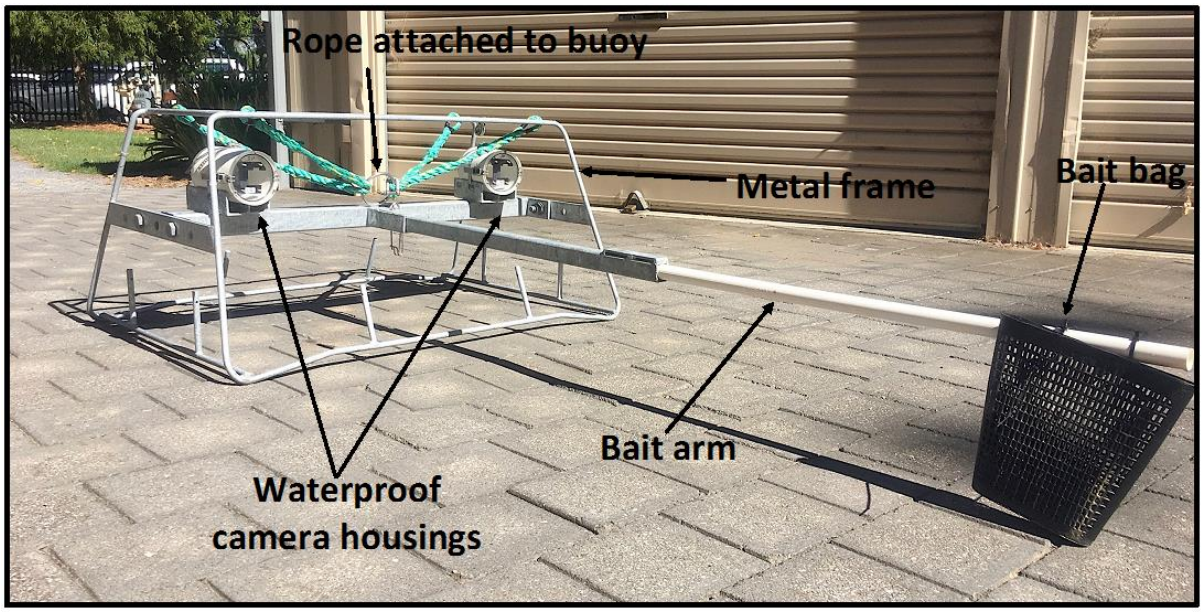


Figure 3. BRUV unit. Frame constructed of aluminium; bait arm was constructed of PVC pope and bait pouch gutter mesh; ropes were connected to a metal ring on one end and the other end was connected to a buoy which allowed the unit to be deployed and retrieved remotely; GoPro video cameras were mounted inside the underwater housings.

DATA ANALYSIS

Videos were downloaded and analysed using SeaGIS' EventMeasure, an event logging and 3D measuring software package designed specifically for biological information and animal behaviour in underwater movie sequences (www.seagis.com.au). For each species on each video, the maximum number of individuals observed in a single frame (*maxN*) was recorded (see Appendix A). *MaxN* is a conservative measure of relative density, which avoids recounting of particular individuals which may revisit the bait. From each *maxN* frame all fish species were measured using the 3D length measurement tool. The average fish size of each species was then calculated to determine differences in fish size between sites. Species richness was recorded by identifying the total number of fish species between sites. Species were identified using online and hard copy resources (Hutchins and Swainston 1986; Gomon *et al.* 2008). The settings used on EventMeasure were standard practices of the Department of Environment Water and Natural Resources (DEWNR) (see Appendix B). Depth was recorded from the vessel's depth sounder, soak time was standardised at 60 minutes, and the field of view, bias and visability were calculated following DEWNR's standard practices (see Appendix B). Habitat was categorised as: seagrass, macro algae, sand, sponge, or broken sand. Broken sand was defined as any habitat which contained around half sand and half seagrass or seaweed. Species of questionable identification were flagged and viewed again, with the assistance of colleagues where necessary. Footage was viewed by a single observer to avoid any variation between viewers.

STATISTICAL ANALYSIS

ABUNDANCE AND DIVERSITY ANALYSIS

Using PRIMER software Version 6 (http://www.primer-e.com/), abundance and species richness data were square root transformed and fitted to a Bray-Curtis dissimilarity resemblance matrix (Bray and Curtis 1957). The square root transformation was applied to avoid dominance of common species and allow contribution from the rarer species. Sites (Coffin Bay and Venus Bay) were analysed both individually (e.g. inside sanctuary zone versus outside sanctuary zone) and combined. Analyses used single factor (inside versus outside sanctuary zone) permutational univariate analyses of variance (ANOVA). For all tests 9999 unrestricted permutations and Monte Carlo simulations were performed. Similar analyses were also undertaken on dominant species and genera.

ASSEMBLAGE ANALYSES

Fish assemblage data from Venus Bay, Coffin Bay and both bays combined were also square root transformed and fitted into a Bray-Curtis dissimilarity resemblance matrix. Venus Bay, Coffin Bay and combined location fish assemblage data were analysed for differences between inside and outside sanctuary zones using single factor permutational multivariate analysis of variance (MANOVA). Additional analyses also investigated differences between sanctuary zone location, depths, season and habitats. For all tests, 9999 unrestricted permutations and Monte Carlo simulations were performed.

Post hoc pairwise tests were conducted to determine where significant differences occurred. For the fish assemblage data, Multi-Dimensional Scaling (MDS) was used to assess both dissimilarity between deployments and identify factors causing these dissimilarities (Clarke 1993). Vectors of species were overlaid on each plot to show the species that contributed to the dissimilarities between sites. These were identified using between group similarities (SIMPER) (Clarke 1993). The stress value was used as an indication of how well the similarity matrix was represented by the non-metric multidimensional scaling plot, where stress levels are closest to zero when the data are perfectly represented (Clarke 1993).

LENGTH ANALYSIS

Dominant species and genera were used to compare length frequency differences between inside and outside sanctuary zones. A Kolmogorov-Smirnov test was run on the frequency data to test if the two distributions differed, and hence quantify their statistical strength.

RESULTS

In total, 498 individuals from 17 species of finfish were identified from 39 BRUV deployments in 2017 (Table 2-3). The most commonly occurring species were *Arripis georgianus*, *A. truttaceus* and *Sillaginodes punctata* (Table 3). During January, 299 individuals from 14 species were identified in 14 BRUV deployments. The June sampling found 199 individuals from 11 species in 25 BRUV deployments (see Appendix C). Temperature differences occurred between these two periods with the average temperature in January being 23.04°C and the average temperature in June being 13.42°C.

Table 2. Summary of information relating to sampling sites and environmental data for January and June sampling. Sampling information includes: sanctuary zone, whether sampling was inside (I) or outside (O) the sanctuary zone, number of BRUV deployments, average depth, and average temperature and dominant habitat.

	Sanctuary zone	Inside/ Outside	Number of BRUV deployments	Average depth (m)	Average temp (ºC)	Habitat
January						
Coffin Bay	1	I	2	4.30	21.97	Seaweed
Coffin Bay	1	O	1	6.30	22.63	Broken sand
Coffin Bay	3	I	2	1.45	23.45	Seagrass
Coffin Bay	3	O	2	2.30	23.68	Seagrass
Coffin Bay	4	I	0	-	-	-
Coffin Bay	4	O	2	4.60	23.14	Seagrass, sand
Venus Bay	8	I	2	1.95	22.72	Seagrass
Venus Bay	8	O	3	1.87	23.34	Seagrass
June						
Coffin Bay	1	I	4	3.25	13.41	Seagrass, sand, sponge
Coffin Bay	1	O	4	4.60	13.38	Seagrass, sand
Coffin Bay	3	I	2	1.60	12.76	Seagrass
Coffin Bay	3	O	2	2.25	12.87	Seagrass
Coffin Bay	4	I	1	5.00	13.32	Sand
Coffin Bay	4	O	2	5.40	13.74	Sand, broken sand
Coffin Bay	5	I	2	1.60	13.20	Seagrass
Coffin Bay	5	O	2	2.50	13.42	Seagrass
Venus Bay	8	I	4	2.15	13.97	Seagrass
Venus Bay	8	O	2	1.75	13.61	Seagrass

Table 3. Summary of fish species identified from BRUV footage analysed. The number of deployments the species were found in is also indicated for each bay. Species codes indicated are used in graphical results.

Species name	Code	Number of deployments species were found in	
		VENUS BAY (N=11)	COFFIN BAY (N=28)
Arripis georgianus	Arrgeo	6	7
Arripis truttaceus	Arrtrut	6	14
Pseudocaranx wrightii	Pseuwri	5	1
Sillaginodes punctata	Sillpun	9	8
Pelates octolineatus	Peloct	3	3
Sillago schomburgkii	Sillscho	1	2
Sphyraena forsteri	Sphfor	1	0
Mustelus antarcticus	Musant	1	0
Sphyraena novaehollandiae	Sphnov	1	0
Platycephalus spp.	Plat	0	11
Acanthaluteres vittiger	Acanvit	0	4
Upeneichthys vlamingii	Upenvla	0	1
Trachurus novaezelandiae	Tracnov	0	1
Genypterus tigerinus	Genytig	0	1
Atule mate	Atumat	0	1
Notolabrus parilus	Notopar	0	2
Nelusetta ayraudi	Nelayr	0	1

ABUNDANCE AND DIVERSITY ANALYSIS

I. VARIATION BETWEEN INSIDE AND OUTSIDE SANCTUARY ZONES FOR LOCATIONS

Total abundance was slightly higher inside sanctuary zones compared to outside for both Coffin Bay and Venus Bay locations, although no significant differences were found (P=0.842, P=0.904) (Figure 4, Table 4). The difference between fish abundance inside and outside sanctuary zones during January sampling was higher than during June sampling (Figure 5, Table 4). This difference was particularly evident in the Coffin Bay January abundance (p=0.056) (Table 4). There were no significant differences for other response variables (species richness, seasonal abundance) (Figure 4-5, Table 4-5).

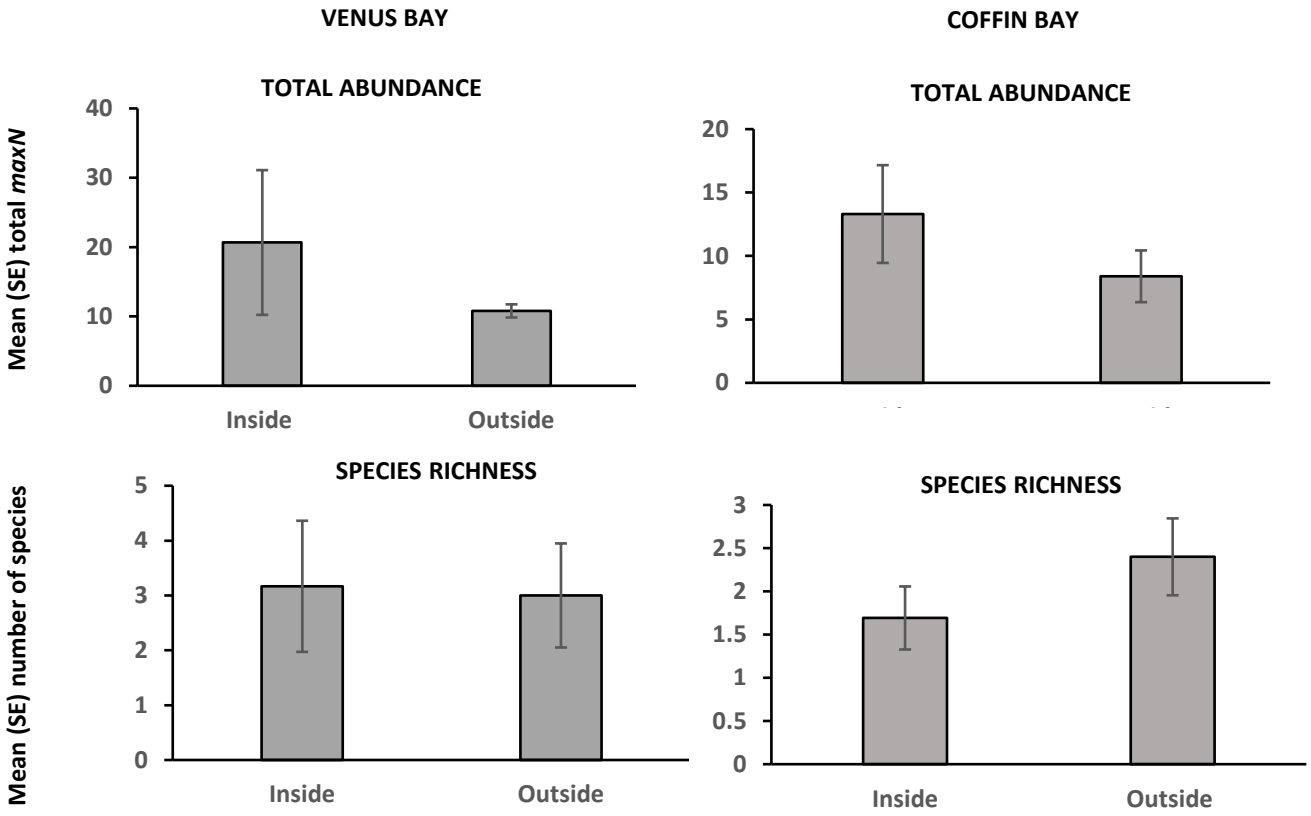


Figure 4. Mean (± SE) total *maxN* and species richness of fish assemblages in Coffin Bay and Venus Bay inside and outside sanctuary zones.

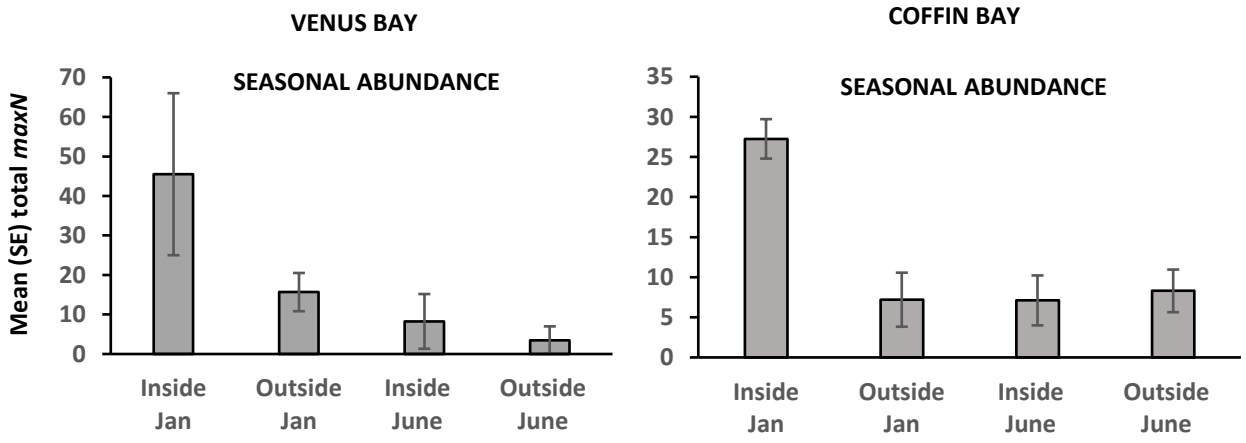


Figure 5. Mean (± SE) total *maxN* for fish assemblages inside and outside sanctuary zones in Coffin Bay and Venus Bay in January and June 2017.

Table 4. Single-factor permutational ANOVA results comparing fish abundance inside and outside sanctuary zones in Coffin and Venus Bays.

SITE	MODEL	Df	MS	F	P
Total abundance (both locations)	Inside/Outside zone	1	27.285	1.040	0.845
	Residual	37	354.980		
Coffin Bay total abundance	Inside/Outside zone	1	17.482	0.720	0.842
	Residual	26	328.710		
Venus Bay total abundance	Inside/Outside zone	1	7.241	0.194	0.904
	Residual	9	504.180		
Coffin Bay January abundance	Inside/Outside zone	1	2043.4	3.883	0.056
	Residual	7	526.27		
Venus Bay January abundance	Inside/Outside zone	1	163.190	2.046	0.292
	Residual	3	79.770		
Coffin Bay June abundance	Inside/Outside zone	1	237.120	0.726	0.366
	Residual	17	326.410		
Venus Bay June abundance	Inside/Outside zone	1	133.780	0.193	0.862
	Residual	4	692.430		

Table 5. Single-factor permutational ANOVA results comparing number of species inside and outside sanctuary zones in Coffin and Venus Bays.

SITE	MODEL	Df	MS	F	P
Total species richness	Inside/Outside zone	1	25.675	0.119	0.733
	Residual	37	215.330		
Coffin Bay total species richness	Inside/Outside zone	1	45.289	0.229	0.657
	Residual	26	197.640		
Venus Bay total species richness	Inside/Outside zone	1	1.085	0.174	1.000
	Residual	9	309.830		
Coffin Bay January species richness	Inside/Outside zone	1	58.630	0.445	0.892
	Residual	7	192.330		
Venus Bay January species richness	Inside/Outside zone	1	30.497	2.064	0.614
	Residual	3	14.775		
Coffin Bay June species richness	Inside/Outside zone	1	188.930	0.913	0.280
	Residual	17	206.890		
Venus Bay June species richness	Inside/Outside zone	1	105.870	0.244	0.863
	Residual	4	433.720		

II. VARIATION BETWEEN INSIDE AND OUTSIDE SANCTUARY ZONES FOR SPECIES

Certain dominant species and genus groups showed trends of having higher abundances inside sanctuary zones, although none of the differences were significant (Figure 6, Table 6). Both *A. georgianus* and *A. truttaceus* had higher abundances inside sanctuary zones at both locations (Figure 6), however there was considerable variability both inside and outside sanctuary areas. Similarly, *P. wrightii* and *S. punctata* both had higher abundances inside the sanctuary zone in Venus Bay (Figure 6). The only dominant genus that had higher abundance outside the sanctuary zone was *Platycephalus* spp. in Coffin Bay (Figure 6).

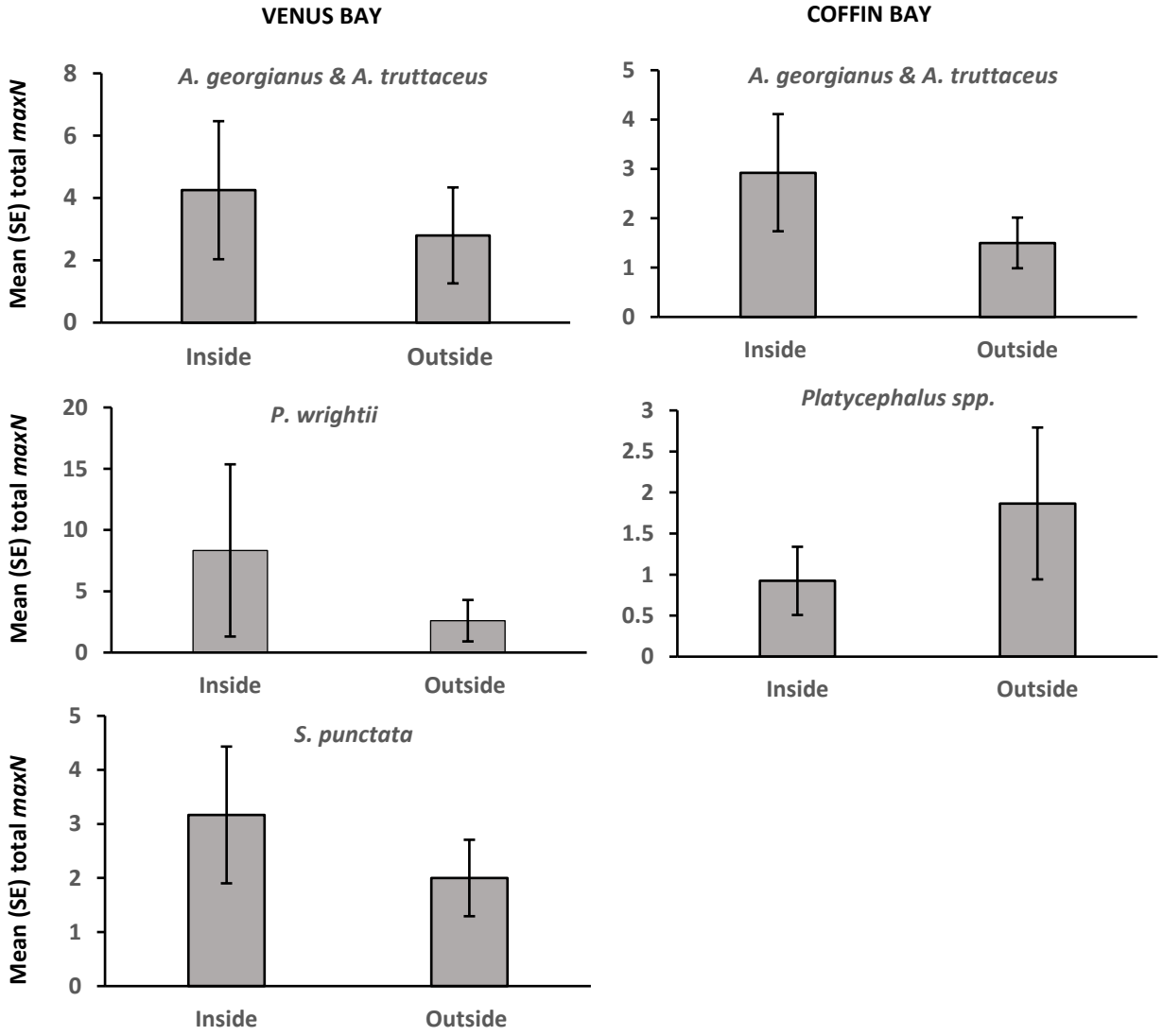


Figure 6. Mean (± SE) total *maxN* of numerically-dominant genera groups and species inside and outside sanctuary zones in Venus Bay and Coffin Bay. Only highly abundant species are included.

Table 6. Single-factor permutational ANOVA results comparing fish abundances inside and outside sanctuary zones in Coffin Bay and Venus Bay for numerically-dominant genera and species.

SITE	MODEL	Df	MS	F	P
Venus bay *A. truttaceus and A. georgianus*	Inside/outside zone	1	321.19	0.2956	0.6093
	Residual	9	1086.7		
Coffin bay *A. truttaceus and A. georgianus*	Inside/outside zone	1	319.79	0.4048	0.5657
	Residual	26	789.88		
Venus bay *P. wrightii*	Inside/outside zone	1	306.6	0.2946	0.7079
	Residual	9	1048.8		
Coffin bay *Platycephalus* spp.	Inside/outside zone	1	32.489	0.7887	0.9116
	Residual	26	557.52		
Venus bay *S. punctata*	Inside/outside zone	1	56.962	0.1322	0.7263
	Residual	9	430.85		

ASSEMBLAGE ANALYSIS

I. ASSEMBLAGE VARIATION BETWEEN INSIDE AND OUTSIDE SANCTUARY ZONES

The structure of fish assemblages between inside and outside sanctuary zones at both locations showed no significant differences (P=0.9330) (Figure 7, Table 7).

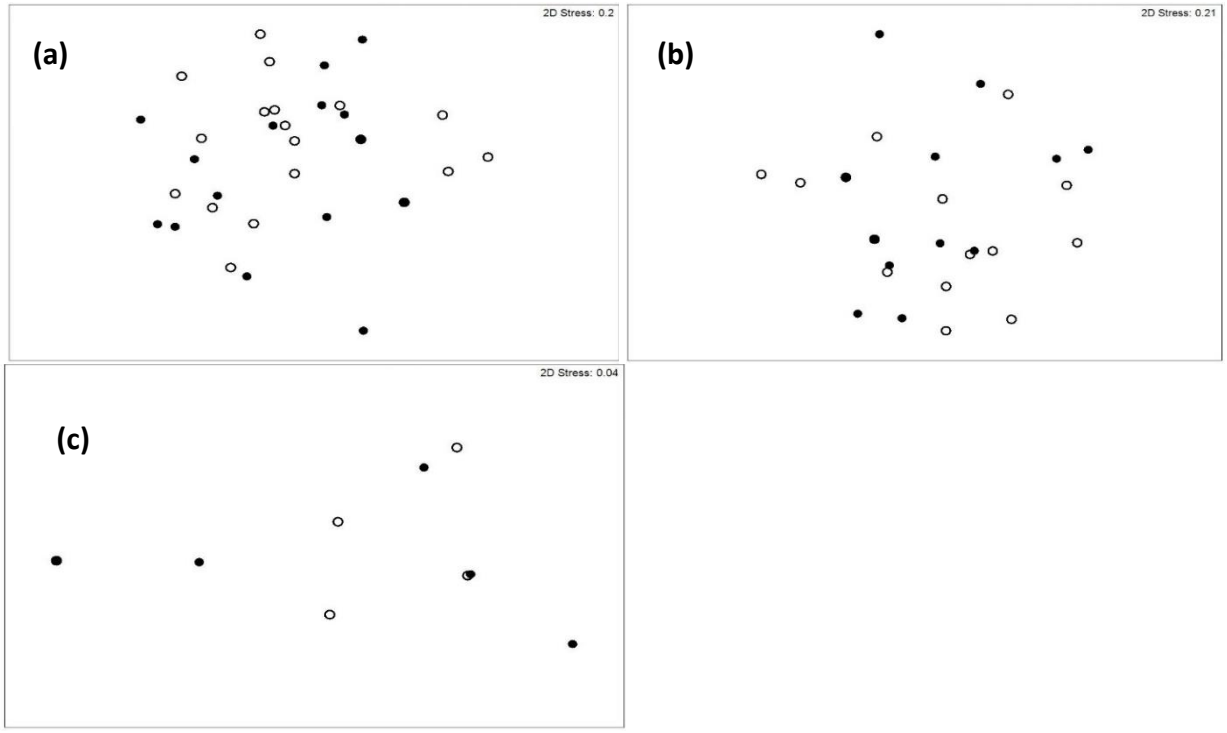


Figure 7. MDS ordination of fish assemblages represented as centroids for each site within sanctuary zone (black circles) and outside sanctuary zone (white circles), where: (a) is combined locations, (b) is Coffin Bay, and (c) is Venus Bay. Fish assemblages are Bray-Curtis similarity measures following square root transformations.

SITE	MODEL	Df	MS	F	P
Combined	Inside/Outside zone	1	642.69	0.2930	0.9330
	Residual	37	2197.20		
Coffin bay assemblage	Inside/Outside zone	1	71.48	0.5210	0.9790
	Residual	9	1855.80		
Venus bay assemblage	Inside/Outside zone	1	819.19	0.3640	0.9040
	Residual	26	2252.70		

Table 7. Single factor permutational MANOVA results comparing fish assemblages inside and outside sanctuary zones in Coffin Bay and Venus Bay. Combined represents both Coffin Bay and Venus Bay data.

II. FACTORS CONTRIBUTING TO ASSEMBLAGE VARIATION

The structure of fish assemblages at both locations was significantly different depending on environmental factors and geographic locations. Fish assemblages varied by depth ($F_{5,33}$ =2.371, P=0.0004) (Figure 8, Table 8). *Post hoc* pairwise analysis revealed fish assemblages differed between some but not all the depths (Appendix D). Most of the significant differences were between the larger depth differences, e.g. between 1m and 6m (P=0.007) (Appendix D).

Fish assemblage structure also varied by habitat ($F_{4,34}$=1.918, P=0.0041) (Figure 9, Table 8). *Post hoc* pairwise analysis showed that significant differences occurred only between seagrass and broken sand habitats (P=0.0108) (Appendix D).

There were further significant differences in fish assemblages between January and June sampling ($F_{1,37}$=6.256, P=0.0002) (Figure 10, Table 8). Although insignificant, trends were also seen for *A. georgianus, P. wrightii, S. punctata* and *P. octolineatus*, which all had higher abundances in January than in June. Several other groups (*A. truttaceus* and *Platycephalus* spp.) had higher abundances in June than in January.

The different sanctuary zone sites also showed significant differences in fish assemblages ($F_{4,34}$=4.564, P=0.0001) (Figure 11, Table 8). *Post hoc* pairwise analysis showed significant differences between all sanctuary zones except for sanctuary zone 8 (Venus Bay) and sanctuary zone 3 (Coffin Bay) (Appendix D).

The species which contributed most to the assemblage differences for all factors were *A. truttaceus, A. georgianus, P. wrightii, S. punctata* and *P. octolineatus* (Figure 8-11).

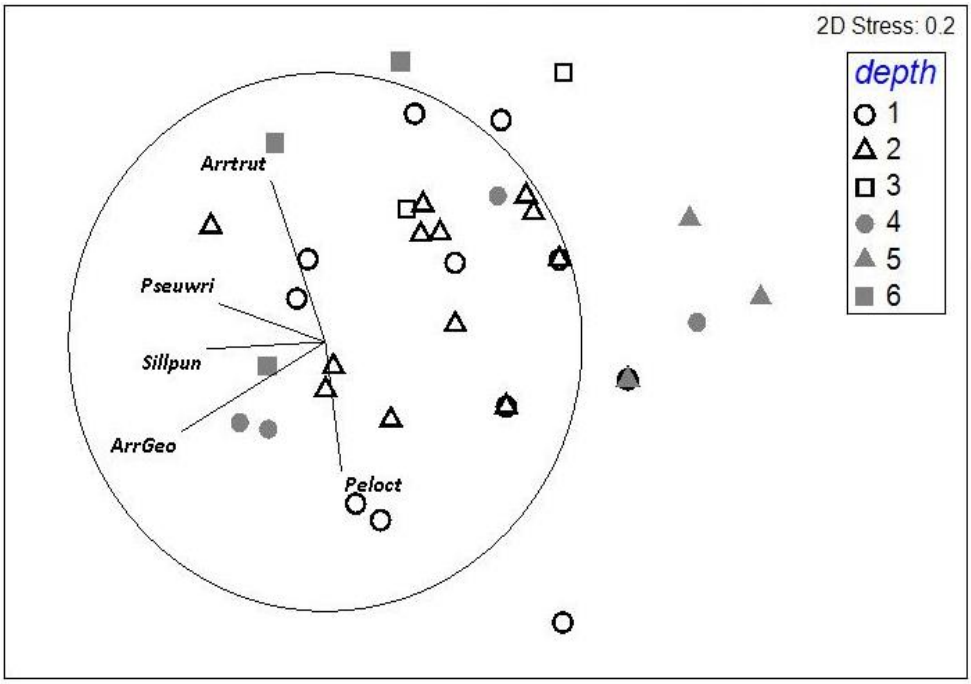


Figure 8. MDS ordination of fish assemblages from both Coffin Bay and Venus Bay separated into depth bins of one metre. Overlay vectors of species contributing to differences among depths (codes are shown in Table 3). Fish assemblages used Bray-Curtis similarity measures following square root transformations. The stress value is shown on the plot.

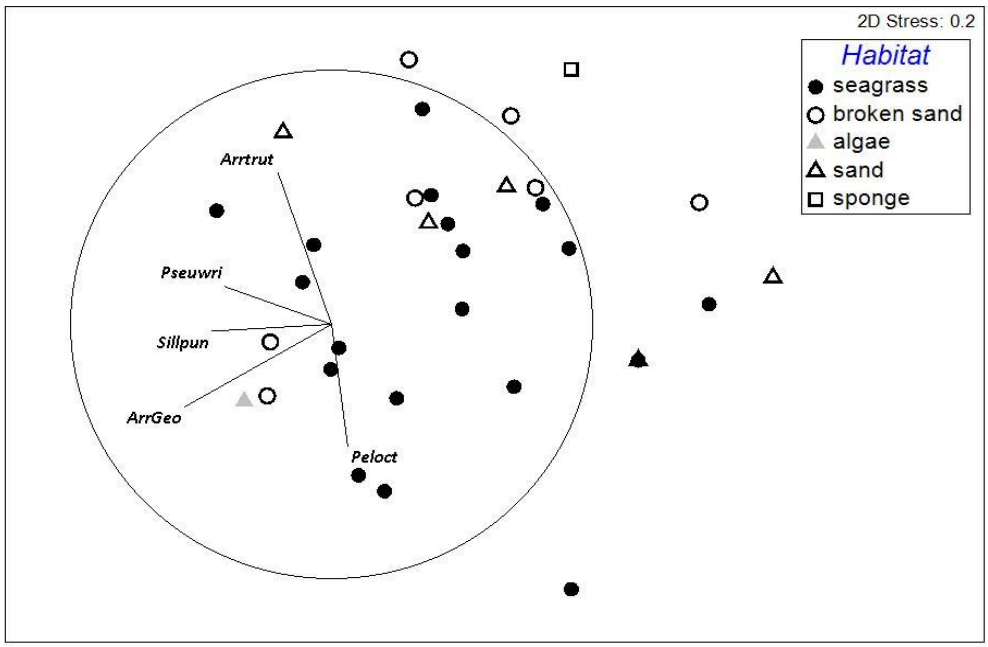


Figure 9. MDS ordination of fish assemblages from both Coffin Bay and Venus Bay separated by habitat type. Overlay vectors of species contributing to differences in sites (codes are shown in Table 3). Fish assemblages used Bray-Curtis similarity measures following square root transformations. The stress value is shown on the plot.

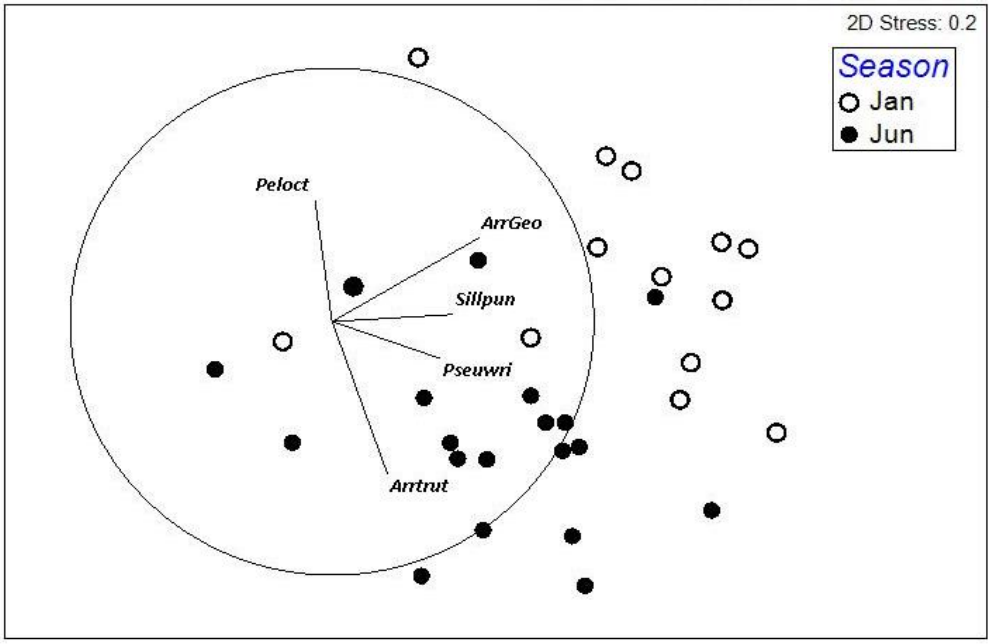


Figure 10. MDS ordination of fish assemblages from both Coffin Bay and Venus Bay separated by sampling time (January and June). Overlay vectors of species contributing to differences between seasons (codes are shown in Table 3). Fish assemblages used Bray-Curtis similarity measures following square root transformations. The stress value is shown on the plot.

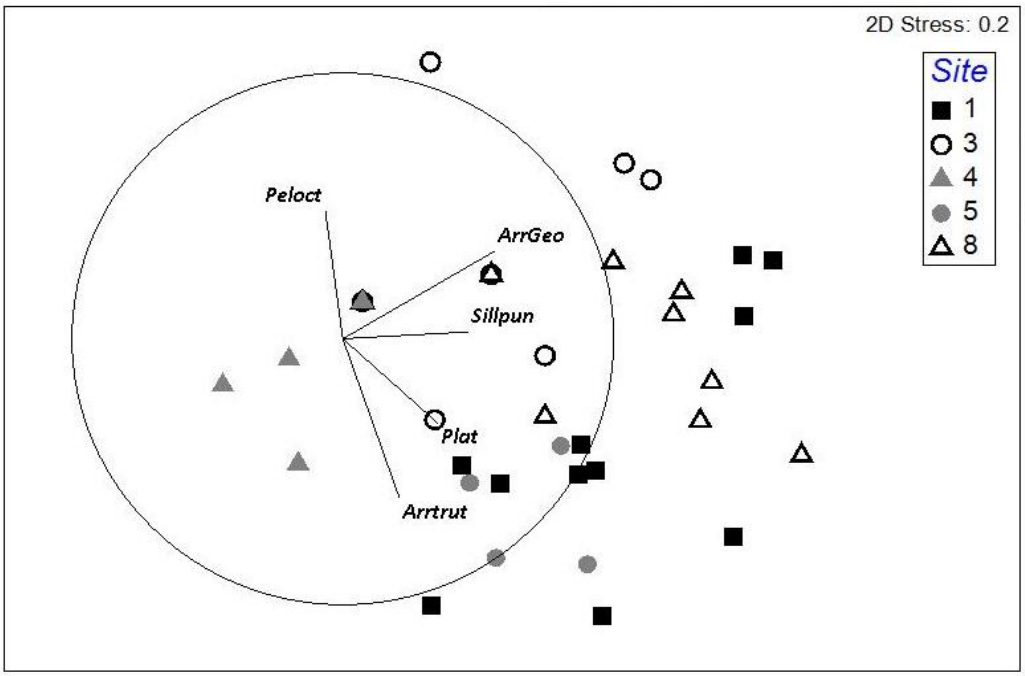


Figure 11. MDS ordination of fish assemblages from both Coffin Bay and Venus Bay separated by sanctuary area. Sanctuary areas 1, 3, 4 and 5 are located in Coffin Bay and sanctuary area 8 in Venus Bay. Overlay vector of species contributing to differences in sites (codes are shown in Table 3). Fish assemblages used Bray-Curtis similarity measures following square root transformations.

Table 8. Single factor permutational MANOVA results comparing fish assemblages with different factors (depth, habitat, season and sanctuary zone (by location)).

FACTOR	MODEL	Df	MS	F	P
Depth	1m, 2m, 3m, 4m, 5m, 6m	5	4332.00	2.371	**0.0004**
	Residual	33	1826.60		
Habitat	Seagrass/broken	4	3771.30	1.918	**0.0041**
	Residual	34	1966.30		
Season	January/June	1	11851.00	6.256	**0.0002**
	Residual	37	1894.30		
Sanctuary zone (by location)	1/3/4/5/8	4	7156.90	4.564	**0.0001**
	Residual	34	1568.00		

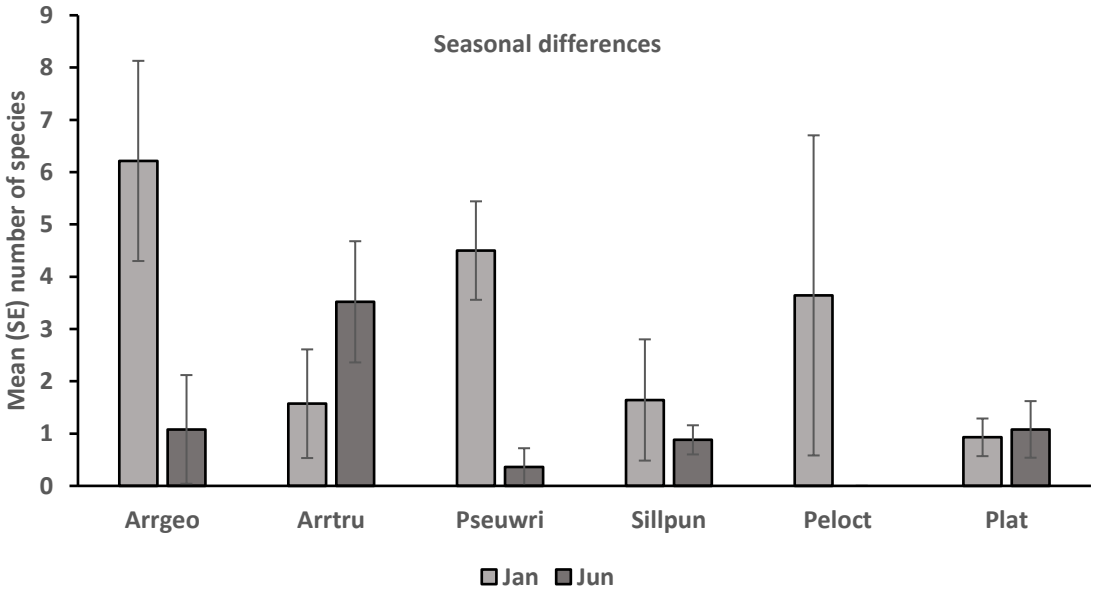


Figure 12. Mean number of numerically-dominant species and genus groups in January (light grey) and June (dark grey) sampling periods.

LENGTH ANALYSIS

The length frequency of dominant species and genus groups varied depending on location (either Coffin Bay or Venus Bay) and species, although no differences in frequencies were significant (Figure 13, Table 9). Generally, there was a trend of a higher percentage of smaller fish inside sanctuary zones (Figure 13). This is evident for *Platycephalus* spp. in Coffin Bay, *A. georgianus* in both Coffin Bay and Venus Bay, *A. truttaceus* in Coffin Bay, and *S. punctata* in both locations (Figure 13).

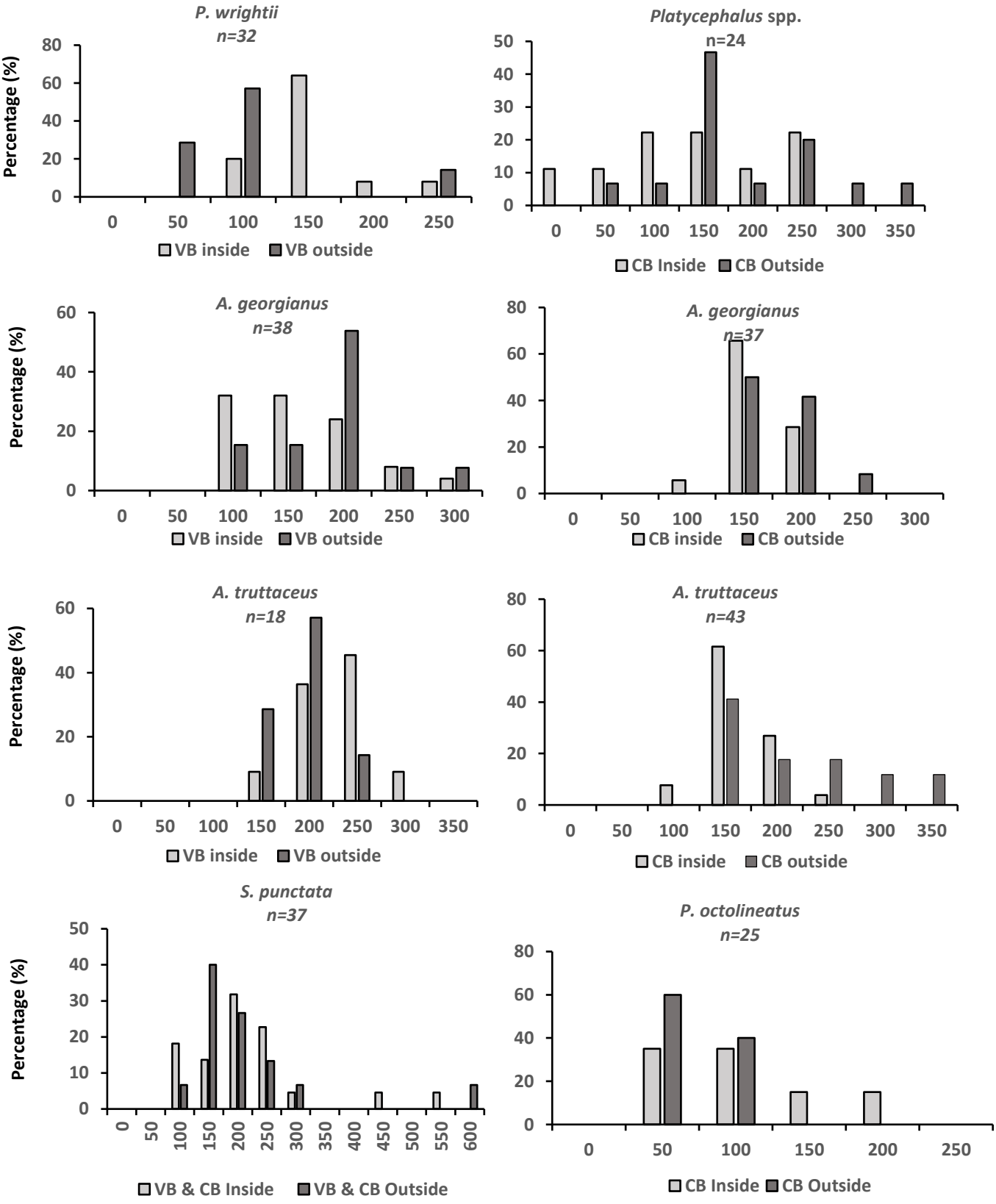


Figure 13. Length frequency of numerically-dominant species and genus groups inside (light grey) and outside (dark grey) sanctuary zones in Coffin Bay and Venus Bay. Where n is the sample size.

Table 9. Kolmogorov-Smirnov values and p-values of the differences in distribution of the length data between inside and outside the sanctuary zones.

Location	Species/genus	Kolmogorov-Smirnov value	P-value
Venus Bay	*Pseudocaranx wrightii*	0.167	1.000
Coffin Bay	*Platycephalus* spp.	0.500	0.270
Venus Bay	*Arripis georgianus*	0.286	0.938
Coffin Bay	*Arripis georgianus*	0.143	1.000
Venus Bay	*Arripis truttaceus*	0.125	1.000
Coffin Bay	*Arripis truttaceus*	0.375	0.627
Venus Bay	*Sillaginodes punctata*	0.154	0.998
Coffin Bay	*Pelates octolineatus*	0.333	0.893

DISCUSSION

EFFECTIVENESS OF SANCTUARY ZONES

South Australia's recently developed sanctuary zones within marine parks are showing promising signs of increasing fish abundances and diversity. There were no significant differences between abundance, species richness and size of specific fish species between inside and outside sanctuary zones. Although not statistically significant, there were still clear trends showing that sanctuary zones in both Venus Bay and Coffin Bay have higher total fish abundances inside rather than outside. When combined, there were 76% more fish in terms of abundance inside the sanctuary zones compared to outside. In a broad sense, these are positive signs for the potential future success of these sanctuary zones.

When abundance and diversity data were separated by location, there were no significant differences inside Sanctuary Zone 8, the only underwater sanctuary zone in Venus Bay, or the four sanctuary zones analysed in Coffin Bay. Despite the recent establishment of South Australia's protected areas, there were positive trends in both locations' sanctuary zones. There were 90% more fish inside the Venus Bay sanctuary zone compared to outside, and 58% more fish inside the Coffin Bay sanctuary zones. In both locations, the key abundance difference occurred in January. In Venus Bay there was almost three times the amount of fish inside the sanctuary zone compared to outside. In Coffin Bay there were 278% more fish inside protected zones, with the significantly lower p-value most likely a result of less variation and a higher sampling size. These results support that these sanctuary zones are functioning as they should, and are particularly encouraging considering their recent establishment.

Despite abundance showing positive signs of increasing, there were no differences in the species richness of fishes between inside and outside sanctuary zones at both locations. This suggests specific species are the cause of high differences in abundance data. Within Venus Bay, four main species dominated, including *A. georgianus*, *A. truttaceus*, *P. wrightii* and *S. punctata.* These four all have higher average abundances inside Sanctuary Zone 8 compared to outside, although none of them were found to be significant. The large variation within the abundance data may have contributed to the lack of significance. For example, although there were 220% more *P. wrightii* inside Sanctuary Zone 8 than in fished areas, the comparison

was not significant due to high amounts of variation of total abundance between inside and outside. This could be in part due to *P. wrightii* being a schooling species, meaning that if a large school swam past the BRUV the data could be unrealistically high. The low degrees of freedom due to lack of replicates in this analysis may also reduce power to detect differences.

Fish assemblages within sanctuary zones do not differ from areas outside sanctuary zones at both locations. Fish assemblages are biotic indicators of overall ecosystem health and productivity (Bell 1983). Over time, it is expected that protected areas will improve in environmental health by increasing the structural complexity of habitats, which in turn will cause an increase in fish abundance and richness (Bell 1983). These results suggest the recent establishment of parks has not yet allowed sufficient time for the ocean floor habitat to improve to a level where fish assemblages are differing. Furthermore, the difference in assemblage structure may not be evident due to the close proximity between BRUV deployments, and the similarities in habitats and depths between inside and outside zones.

Protected areas of ocean, particularly in the form of sanctuary zones, are integral to the health and maintenance of natural ocean ecosystems. Numerous studies conclude that MPAs contain higher fish diversity and abundance, with greater average fish size and general higher biodiversity than areas outside MPAs (Russ *et al.* 2008; Kelaher *et al.* 2014; Soler *et al.* 2015). Despite this, it was still expected that the results from this study would not show drastic improvements inside sanctuary zones due to the enforcement of the South Australian MPA's zoning only commencing recently, in October 2014. Other studies investigating similar concepts in recently established MPAs have found that a three year period since reserve establishment may not be sufficient in generating clear cut trends in fish population recoveries (Edgar and Barrett 2012; Kelaher *et al.* 2014). Furthermore, a global meta-analysis suggested that the positive effects of MPA success are linked to the amount of time that has passed since the MPA was established, with reserves established for more than ten years having higher species diversity and abundance (Claudet *et al.* 2008; Edgar *et al.* 2014).

MARINE PROTECTED AREA DESIGN

When interpreting positive abundance trends within MPAs, it is important to consider if the protected areas were purposefully located in areas where fish naturally occur in higher numbers (Kelaher *et al.* 2014). The West Coast Bays Marine Park, of which Venus Bay is a part, and the Thorny Passage Marine Park of which Coffin Bay is a part, were created in

areas according to the guidelines recommended by DEWNR (Baker 2004). The development of the MPAs used existing data, resources and knowledge to better understand the marine habitats and hence protected areas were placed in ecologically important locations (Baker 2004). The recommendations also state that modifications of the existing network of MPAs may occur as the knowledge on the functioning, distribution and environmental impacts of South Australia's unique marine biota is broadened (Department of Environment, Water and Natural Resources 2012). One of the most effective ways to monitor the effectiveness of the protection is to sample the abundances of particular key species over time. Hence, this study provides baseline data which can be built on in future years to continue the monitoring in these regions. In turn, this information can provide rare knowledge on these regions, potentially influencing the design and locations of MPAs and their zoning in the west coast embayment regions.

While individual MPAs provide some conservation benefits, the design of a MPA is also essential to its effectiveness. Previous research has shown that the five key features: enforcement of regulations, full protection, large size (>100km^2), longevity (>10 years since setup), and isolation, are essential for MPA success (Edgar *et al.* 2014; Halpern 2014). Although it is hard to monitor the enforcement of west coast MPAs, research shows that protected areas with boundaries that exist only in principle but have limited enforcement dramatically lessen the MPAs success (Mora *et al.* 2006; Guidetti *et al.* 2008). Furthermore, areas which are only partially protected, such as all of the MPAs in the west coast embayments excluding the sanctuary zones, have significantly less effectiveness than no-take regions (Edgar *et al.* 2014). South Australia's MPAs cover an area of 26,655 square kilometres, although some of the sanctuary zones within, including those tested in this study, are less than the 100km^2 needed for success (Department of Environment, Water and Natural Resources 2012). These factors are important when designing MPAs in the future and ensuring as many as possible are included is key to the future success of the protected areas.

Understanding the distribution of fish assemblages at a wide range of different spatial scales is an essential step towards discovering important underlying ecological processes and factors that affect fish assemblages. This knowledge, in turn, can be used for the selection and design of MPAs to ensure that the appropriate areas are covered. This includes taking environmental

and biological factors into consideration, so that the ways in which they are interacting can be accounted for.

Environmental factors influencing fish assemblages

The results showed that fish assemblages significantly differed between different habitats and depths, and particularly seasonally. Species of fish associated with temperate zones have a wide range of biological characteristics that affect the way in which they respond to the environment, hence their variation in spatial distribution. Particular species prefer to live in certain depths, habitats and migrate and recruit in different seasons. It is for these reasons that it is important to cover a wide range of depths and habitats when designing sanctuary zones within MPAs. Although important in MPA success, the range of depths and habitats raise issues relating to variability within the data and make it is hard to distinguish between the factors which are causing these assemblage differences. Despite this, there are still some obvious fish assemblage differences between environmental factors.

Depth

Significant fish assemblage differences occurred between different depths. Although, this was expected between shallower and deeper areas (Friedlander and Parrish 1998; Connell and Lincoln-Smith 1999; Hyndes *et al.* 1999), in this study depths only varied between shallow zones (one to six metres), so these results were unexpected. Because of similarities in depth range, it is hard to know if the results were in fact related to depth, or rather a spatial difference associated with marine sanctuary zones. For example, in the samples taken inside and outside Sanctuary Zone 1, all of the BRUV deployments were in the 4-6 metre range, while in Sanctuary Zone 8, BRUV deployments were all in 1-2 metres. Furthermore, certain depths were more abundant than other depths, which may have skewed the significance of the data slightly. Despite this, there were assemblage differences across depth ranges, which warrants further investigation. As such, the study design could be modified in the future to distinguish spatial (sanctuary zone) and depth related patterns in fish assemblages.

Habitat

As well as assemblage differences between depths there were also significant differences between different habitats. This was particularly evident between seagrass and sand based habitats. Seagrass is known to act as important habitat for juvenile fish (Aaron *et al.* 2006; Nagelkerken *et al.* 2012), As such, these results are most likely attributable to juvenile fish

species being within the seagrass regions. Habitats are being used more commonly in MPA planning as they are useful surrogates for biodiversity (Ward *et al.* 1999; Harman *et al.* 2003). When planning the design of an MPA, including a variety of habitats is recommended, with all habitats represented within the protected area (Kelleher and Kenchington 1991; Roberts *et al.* 2000). This is beneficial to preserving both the habitats themselves, and the fish that use them (Rosenberg *et al.* 2000).

As mentioned above there are some sampling issues, most likely due to lack of haphazard spacing among sites, therefore results could be due to spatial variation. The results showed that most sanctuary zones had a particular habitat that dominated the region. Sanctuary Zones 3, 5 and 8, for example, had all BRUV deployments in seagrass. Therefore, the results could be due to differences in sanctuary zones mentioned below, rather than differences in habitats. Despite this, there were still significant differences between these seagrass-based sanctuary zones, indicating that spatial patterns are greater than habitat patterns. Furthermore, the dominance of seagrass as a habitat, and the lack of other habitat samples such as macro algae, could have caused some issues with statistical power of the analysis. The statistical power of an analysis is increased with sample size, hence in our study where there is just one site with macro algae, the sample size is very low for this particular analysis. Some changes to the study design could be incorporated to increase the statistical power, such as ensuring that each sanctuary zone has a wide variety of habitats and depths tested. Despite this, the differences between fish assemblages among habitats are interesting, and supported by a wide range of literature (Guidetti 2000; Gratwicke and Speight 2005), and confirm the necessity for sanctuary zones to be located in a range of habitats.

SEASON

Seasonally, fish assemblages are expected to undergo changes, cycling consistently among years (Wright 1988; Hyndes *et al.* 1999). The results from this study showed clear differences in fish assemblages between January and June sampling periods. The seasonal differences may be attributable to immigration and emigration of different fish species including recruitment throughout the year (Ansari *et al.* 1995; Potter *et al.* 1997). Particular nursery species could be moving out to deeper waters or fish could be migrating to spawning locations (Hyndes *et al.* 1999). The water temperature differed between these sampling

periods, so lower abundances of fish during winter sampling could also be attributed to temperature changes.

When specific species and genera were investigated there were clear differences in abundances between seasons, which supports the idea that fluctuations relate to the life cycles and emigrations and immigrations of particular species. *P. octolineatus,* for example, was only recorded during January sampling, with 51 individuals sited in January compared to zero in June. This is expected as previous research found that *P. octolineatus* migrate from seagrass nursery areas into deeper waters to mature and spawn during spring (Potter *et al.* 1983; Veale *et al.* 2015). Therefore, by January, it is expected that the abundance of juvenile *P. octolineatus* would be high in seagrass meadows, where most of the sanctuary zones are located. It is likely that other species follow similar life cycles and trends.

SPATIAL VARIATION INFLUENCING FISH ASSEMBLAGES

SANCTUARY ZONE SITE

Variation in fish assemblages at a range of different spatial scales is expected for temperate reef fishes (Anderson and Millar 2004; García-Charton *et al.* 2004; Gladstone 2007). Significant spatial variation in fish assemblages occurred between individual sanctuary zones. This variation occurred at a scale of 1-10 kilometres. Spatial variation on a small scale may be related to variation in structure of the habitat (Connell and Jones 1991; Willis and Anderson 2003), depth, recruitment (Connell and Jones 1991; Smith *et al.* 1991), local larval accumulation and retention (Warner *et al.* 2000), or other influences.

The design and layout of MPAs are often limited due to social and economic factors. This is particularly seen when the size of protected areas is limited to a fraction of the bioregion whose biodiversity they are intended to represent. Often, they are not large enough to be self-sustaining as their size is smaller than the dispersal distance of key species (Halpern 2003; Claudet *et al.* 2008). This relates to potential issues with the design of the sanctuary zones within South Australian MPAs. The results of this study show that over very small distances there is still variation related to species composition. An easy solution would be creating larger reserves, however due to socio-economic pressures this is unlikely to have community support. An alternative is to create more connectivity between the reserves. In theory, this allows each reserve to contribute and receive a sufficient amount of adults and larvae from connected reserves. Furthermore, connectivity between protected areas is important for

population dynamics and genetics of marine organisms (Palumbi 2003; Cowen *et al.* 2006). This would potentially ensure the protection of species and increase the success of sanctuary zones to replenish a wide range of fish species.

Length of fish in sanctuary zones

While increasing the abundance and diversity of fish, individuals of commercially fished species in MPAs are thought to increase in size. Protected areas provide a safe area for these large fish, which are usually crucial for reproduction, the offspring of which often spills over to areas outside the reserve. Furthermore, protected areas also provide a healthy habitat for recruitment and larval export to occur, with strong evidence of community wide benefits flowing from MPAs to unprotected areas (Hilborn *et al.* 2004; Guidetti 2007; Harrison *et al.* 2012a).

Some size differentiation occurred between inside and outside sanctuary zones. In particular, some species showed trends of higher abundances of smaller sized species inside the sanctuary zones, indicating that recruitment and larval export maybe occurring within the protected areas. This is seen in Coffin Bay for *A. truttaceus* and *Platycephalus* spp., and in both bays for *A. georgianus.* Despite the patterns seen within these results, it is important to keep in mind that the sample sizes of all species were quite small, with all species having a total abundance of 43 or less. Furthermore, the sanctuary zones within both embayments had only been enforced for less than three years when the data were collected, which is most likely not enough time for the fish populations and habitats to recover to a detectable level.

Limitations in the present study

Similar to most studies, limitations occurred throughout phases of this study. As mentioned above, there are issues relating to the time frame since the South Australian MPAs were established, and the enforcement of the sanctuary zones within them. Previous studies have indicated that three years is insufficient time to see obvious changes in fish abundance and diversity, as well as habitat recovery (Edgar and Barrett 2012; Kelaher *et al.* 2014). Despite this, the preliminary results of this study are promising and provide important baseline data that can be used in the future to compare how much the MPAs have improved.

There are also some potential limitations relating to the use of Baited Remote Underwater Video (BRUV) in sampling. BRUVs are a relatively new, popular technique which are effective at assessing demersal and nektonic aquatic assemblages, particularly fish (Ellis and DeMartini 1995; Cappo *et al.* 2004; Whitmarsh *et al.* 2017). Stereo-video techniques reduce inter-observer variability, improve the definition of the area sampled, increase the accuracy of fish length data and provide a permanent record of the data (Langlois *et al.* 2010; Gibson *et al.* 2016). Although the permanency of the footage from BRUVs means there is ample opportunities to view earlier data, biases still pose issues (Murphy and Jenkins 2010).

The use of bait means that mainly carnivorous species are attracted, leaving a potential gap in the data relating to herbivorous species (Harvey *et al.* 2007; Hardinge *et al.* 2013). Another concern with the use of bait is that it may preferentially sample particular sizes of fish compared to unbaited methods, due to larger fish needing more food (Hardinge *et al.* 2013; Klages *et al.* 2014). There are complications surrounding the relationship between fish abundance and the size of the bait plume, as variations with environmental influences such as current flow differ between sites (Hardinge *et al.* 2013). Ideally, the bait plume would be estimated and controlled for. Furthermore, the behaviour of specific species can affect their abundance. For example, the large amount of activity that occurs around the bait bag sometimes can mean that shy fish species are not recorded (Priede and Merrett 1998; Bailey and Priede 2002).

Fish assemblages and abundances vary between years, seasons and times of day (Willis *et al.* 2006; Birt *et al.* 2012). Although this study was able to compare seasonal differences, lack of time for the project limited the chance for temporal analyses of both between year and between day scales to occur (Birt *et al.* 2012). There are significant differences between fish assemblages at different times of the day, most likely related to specific fish species activity and feeding times (Birt *et al.* 2012). It would be expected that there would be variations in abundance and species richness on various temporal scales. In this study, all sampling occurred during daylight hours between 8am and 8pm, with samples at any one time being placed both inside and outside MPAs. Future monitoring of South Australian MPAs could investigate the temporal scale at which greatest variation occurs (Morrisey *et al.* 1992). For example, a nested sampling design involving days, weeks, months and seasons could be used.

Due to limited time and resources, the number of BRUV deployments, and hence the sampling replicates, may be too low to detect differences due to lack of power. Ideally, significantly more sample replicates would be taken, inside and outside all of the sanctuary zones. Furthermore, repetition of samples, at similar locations in both seasons would occur. Likewise, the experimental design could be improved by the location of deployments. As mentioned above, there are some issues relating to the results potentially being an effect of spatial variation rather than the factors tested. This occurred due to haphazard spacing among both locations and sites. Testing a range of depths and habitats within each sanctuary zone would mean that a more accurate representation of the factors contributing to fish assemblage differences could be obtained, but would require more BRUV deployments than was possible for this study.

FUTURE DIRECTIONS

The use of BRUVs to analyse the effectiveness of MPAs is a promising technique. However, to enhance the analysis and potentially improve the validity of results, the use of other survey methodologies alongside BRUVs could be beneficial. This could include Underwater Visual Census (UVC), Diver Operated Video (DOV) and angling. A commonly found outcome is that no particular technique is perfect for providing data on all fish species, and a combination of different techniques is usually the most accurate (Willis and Babcock 2000; Murphy and Jenkins 2010). Furthermore, different survey methods suit particular species, depending on the biology and behaviour of the species of interest (Trevor *et al.* 2000).

As mentioned above, a power analysis could be completed to improve the validity of the results. The data collected in this research provides an opportunity to undertake a power analysis to predict the number of replicates required to detect effects, which could then be applied to future research by repeating samples in a range of different habitats and depths. This would help distinguish if spatial differences are in fact due to differences within the sanctuary zone itself, or instead vary due to biological factors. In turn, this could further expand the scientific data needed to make recommendations for ideal MPA and sanctuary zone design. Additionally, the size data could be further developed by identifying a commercially important species, for example King George Whiting (*S. punctata),* and creating an age-growth curve to see how much they grow within a particular temporal period.

In terms of future studies, it would be ideal if all South Australian MPAs could be monitored in the near future to provide an essential set of baseline data, similar to this study. This will need to be completed before the MPAs age and differences between protected and unprotected areas cannot be fully accounted for. Although DEWNR are currently collecting data, there is still potential to further expand this data set, potentially by incorporating citizen science methods. Worldwide, it is increasingly popular to use citizen volunteers to help monitor natural resources, conserve protected areas and observe potential species at risk (Delaney *et al.* 2008; Conrad and Hilchey 2011). This could include collecting abundance data using a standardised visual method while snorkeling or diving (Pattengill-Semmens and Semmens 2003). Alternatively, small, portable BRUV units could be deployed from fisher's boats in a quest to involve potentially the most influential people on MPA success (Pattengill-Semmens and Semmens 2003).

Furthermore, linking the monitoring of MPAs to the local people may help to make the concept of marine protection and conservation more relevant, and hence the MPAs more sustainable (Danielsen *et al.* 2009). The involvement of the community in the success of MPAs, and hence the conservation of unique marine biota, is essential. Communication with coastal communities, by allowing them access to the data collected in studies such as this one, is important. By showing the general public the success of MPAs and the sanctuary zones within them, the likelihood increases of the community accepting them, and following the rules of the 'no-take' concept within the sanctuary zone. A potential option following the completion of this study would be to provide local tourism centres and caravan parks with a short summary of the results, with some simplified pictures and information so that the general public can see the success of MPAs.

CONCLUSION

Marine Protected Areas are increasingly being established as global concern for marine ecosystem health escalates (Halpern *et al.* 2010). MPAs generally contain higher fish diversity and abundance, with greater average size and higher biodiversity, than areas which are unprotected (Kelaher *et al.* 2014; Soler *et al.* 2015). This study provides an essential set of baseline data on fish assemblages, abundance and species richness in South Australia's recently established west coast embayments. Knowledge and baseline data related to MPAs

are essential for the future success and conservation of the unique marine biota within them. Overall, the information provided by BRUVs in this study indicate that there are positive trends occurring within the sanctuary zones in both Coffin Bay and Venus Bay after less than three years of protection. Although not statistically significant, there was still a 76% higher abundance of fish inside sanctuary zones compared to outside, which adds to the growing weight of evidence that protected areas increase fish abundance (Kelaher *et al.* 2014; Soler *et al.* 2015). Fish assemblage analyses showed that not all individual sanctuary areas performed equally, and there were variations between the seasons. Fish assemblages also varied among sanctuary zone locations. This demonstrates the importance of MPA and sanctuary zone design to incorporate a range of different habitats and have some form of connectivity. Potential future applications, such as testing a broad range of different habitats and depths within the sanctuary zones, could be of assistance in the design of future MPAs.

ACKNOWLEDGEMENTS

Enormous thanks to my incredible supervisor Professor Bronwyn Gillanders who was there for me through the entire completion of this thesis, always willing to assist me and direct me when needed. Her amazing combination of professionalism, expertise and kindness has been extraordinary.

Special thanks to Matt McMillan, whose field trip organisation and company was wonderful, and made for a very memorable beginning of Honours. Especially thanks for the ongoing assistance throughout the year, with any questions, troubles or BRUV related problems.

Thanks also to my sister, Hannah, and mum and dad for tirelessly reading this thesis and providing endless advice, love and support throughout the year. Shout out to my friends in Room 104, particularly Jackson Stockbridge, without whose encouragement this thesis would most likely not exist. Further thanks to Emma Westlake, for not only drafting this thesis but also for being my marine biologist idol and providing limitless advice.

Thank you to Dimitri Colella, Danny Brock, Simon Bryars and all the people at DEWNR for the substantial funding opportunities, without which we would not have been able to collect anywhere near the amount of data. Especially thanks to Dimitri, who helped me endlessly with any BRUV related issues and questions.

Further thanks to Nature Foundation SA, whose generous support of a research funding grant assisted greatly in field trip costs and data collection.

REFERENCES

Aaron, JA, Craig, PD, Kellison, GT, Matthew, SK, Craig, AL, Janet, AL, Ivan, N, Joseph, ES (2006) Nursery function of tropical back-reef systems. *Marine Ecology Progress Series* **318**, 287-301.

Anderson, MJ, Millar, RB (2004) Spatial variation and effects of habitat on temperate reef fish assemblages in northeastern New Zealand. *Journal of Experimental Marine Biology and Ecology* **305**, 191-221.

Ansari, ZA, Chatterji, A, Ingole, BS, Sreepada, RA, Rivonkar, CU, Parulekar, AH (1995) Community structure and seasonal variation of an inshore demersal fish community at Goa, West Coast of India. *Estuarine, Coastal and Shelf Science* **41**, 593-610.

Bailey, DM, Priede, IG (2002) Predicting fish behaviour in response to abyssal food falls. *Marine Biology* **141**, 831-840.

Baker, JL, 2004. Towards a system of ecologically representative marine proteced areas in South Australian marine bioregions - technical report. South Australia.

Barr, LM, Possingham, HP (2013) Are outcomes matching policy commitments in Australian marine conservation planning? *Marine Policy* **42**, 39-48.

Barrett, NS, Edgar, GJ, Buxton, CD, Haddon, M (2007) Changes in fish assemblages following 10 years of protection in Tasmanian marine protected areas. *Journal of Experimental Marine Biology and Ecology* **345**, 141-157.

Beck, MW, Heck, KL, Able, KW, Childers, DL, Eggleston, DB, Gillanders, BM, Halpern, B, Hays, CG, Hoshino, K, Minello, TJ, Orth, RJ, Sheridan, PF, Weinstein, MP (2001) The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* **51**, 633-641.

Bell, JD (1983) Effects of depth and marine reserve fishing restrictions on the structure of a rocky reef fish assemblage in the north-western Mediterranean Sea. *Journal of Applied Ecology* **20**, 357-369.

Birt, MJ, Harvey, ES, Langlois, TJ (2012) Within and between day variability in temperate reef fish assemblages: learned response to baited video. *Journal of Experimental Marine Biology and Ecology* **416**, 92-100.

Bray, JR, Curtis, JT (1957) An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs* **27**, 325-349.

Burfeind, DD, Pitt, KA, Connolly, RM, Byers, JE (2013) Performance of non-native species within marine reserves. *Biological Invasions* **15**, 17-28.

Byers, JE, Noonburg, EG (2007) Poaching, enforcement, and the efficacy of marine reserves. *Ecological Applications* **17**, 1851-1856.

Cappo, M, Speare, P, De'Ath, G (2004) Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. *Journal of Experimental Marine Biology and Ecology* **302**, 123-152.

Charton, JAG, Williams, ID, Ruzafa, AP, Milazzo, M, Chemello, R, Marcos, C, Kitsos, MS, Koukouras, A, Riggio, S (2002) Evaluating the ecological effects of Mediterranean marine protected areas: habitat, scale and the natural variability of ecosystems. *Environmental Conservation* **27**, 159-178.

Choat, JH, Ayling, AM (1987) The relationship between habitat structure and fish faunas on New Zealand reefs. *Journal of Experimental Marine Biology and Ecology* **110**, 257-284.

Clarke, KR (1993) Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**, 117-143.

Claudet, J, Osenberg, CW, Benedetti-Cecchi, L, Domenici, P, García-Charton, JA, Pérez-Ruzafa, A, Badalamenti, F, Bayle-Sempere, J, Brito, A, Bulleri, F, Culioli, JM, Dimech, M, Falcón, JM, Guala, I, Milazzo, M, Sánchez-Meca, J, Somerfield, PJ, Stobart, B, Vandeperre, F, Valle, C, Planes, S (2008) Marine reserves: size and age do matter. *Ecology Letters* **11**, 481-489.

Connell, SD, Jones, GP (1991) The influence of habitat complexity on postrecruitment processes in a temperate reef fish population. *Journal of Experimental Marine Biology and Ecology* **151**, 271-294.

Connell, SD, Lincoln-Smith, MP (1999) Depth and the structure of assemblages of demersal fish: experimental trawling along a temperate coast. *Estuarine, Coastal and Shelf Science* **48**, 483-495.

Conrad, CC, Hilchey, KG (2011) A review of citizen science and community-based environmental monitoring: issues and opportunities. *Environmental Monitoring and Assessment* **176**, 273-291.

Cowen, RK, Paris, CB, Srinivasan, A (2006) Scaling of connectivity in marine populations. *Science* **311**, 522-527.

Crain, CM, Halpern, BS, Beck, MW, Kappel, CV (2009) Understanding and managing human threats to the coastal marine environment. *Annals of the New York Academy of Sciences* **1162**, 39-62.

Danielsen, F, Burgess, ND, Balmford, A, Donald, PF, Funder, M, Jones, JPG, Alviola, P, Balete, DS, Blomley, TOM, Brashares, J, Child, B, Enghoff, M, FjeldsÅ, JON, Holt, S, HÜBertz, H, Jensen, AE, Jensen, PM, Massao, J, Mendoza, MM, Ngaga, Y, Poulsen, MK, Rueda, R, Sam, M, Skielboe, T, Stuart-Hill, G, Topp-JØRgensen, E, Yonten, D (2009) Local participation in natural resource monitoring: a characterization of approaches. *Conservation Biology* **23**, 31-42.

Delaney, DG, Sperling, CD, Adams, CS, Leung, B (2008) Marine invasive species: validation of citizen science and implications for national monitoring networks. *Biological Invasions* **10**, 117-128.

Department of Environment, Water and Natural Resources (2012) 'South Australian Marine Parks Network - Explanatory document.' Government of South Australia, South Australia.

Dorenbosch, M, van Riel, MC, Nagelkerken, I, van der Velde, G (2004) The relationship of reef fish densities to the proximity of mangrove and seagrass nurseries. *Estuarine Coastal and Shelf Science* **60**, 37-48.

Edgar, GJ, Barrett, NS (1999) Effects of the declaration of marine reserves on Tasmanian reef fishes, invertebrates and plants. *Journal of Experimental Marine Biology and Ecology* **242**, 107-144.

Edgar, GJ, Barrett, NS (2012) An assessment of population responses of common inshore fishes and invertebrates following declaration of five Australian marine protected areas. *Environmental Conservation* **39**, 271-281.

Edgar, GJ, Stuart-Smith, RD, Willis, TJ, Kininmonth, S, Baker, SC, Banks, S, Barrett, NS, Becerro, MA, Bernard, ATF, Berkhout, J, Buxton, CD, Campbell, SJ, Cooper, AT, Davey, M, Edgar, SC, Forsterra, G, Galvan, DE, Irigoyen, AJ, Kushner, DJ, Moura, R, Parnell, PE, Shears, NT, Soler, G, Strain, EMA, Thomson, RJ (2014) Global conservation outcomes depend on marine protected areas with five key features. *Nature* **506**, 216-220.

Ellis, DM, DeMartini, EE (1995) Evaluation of a video camera technique for indexing abundances of juvenile pink snapper, *Pristipomoides filamentosus*, and other Hawaiian insular shelf fishes. *Fishery Bulletin* **93**, 67-77.

Fitzgerald, DB, Winemiller, KO, Sabaj Pérez, MH, Sousa, LM (2017) Seasonal changes in the assembly mechanisms structuring tropical fish communities. *Ecology* **98**, 21-31.

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

Gaines, SD, White, C, Carr, MH, Palumbi, SR (2010) Designing marine reserve networks for both conservation and fisheries management. *Proceedings of the National Academy of Sciences* **107**, 18286-18293.

García-Charton, JA, Pérez-Ruzafa, Á, Sánchez-Jerez, P, Bayle-Sempere, JT, Reñones, O, Moreno, D (2004) Multi-scale spatial heterogeneity, habitat structure, and the effect of marine reserves on western Mediterranean rocky reef fish assemblages. *Marine Biology* **144**, 161-182.

Gibson, R, Atkinson, R, Gordon, J (2016) A review of underwater stereo-image measurement for marine biology and ecology applications. *Oceanography and marine biology: an annual review* **47**, 257-292.

Gillanders, BM (1997) Patterns of abundance and size structure in the blue groper, *Achoerodus viridis* (Pisces, Labridae): evidence of links between estuaries and coastal reefs. *Environmental Biology of Fishes* **49**, 153-173.

Gladstone, W (2007) Requirements for marine protected areas to conserve the biodiversity of rocky reef fishes. *Aquatic Conservation: Marine and Freshwater Ecosystems* **17**, 71-87.

Gomon, M, Bray, D, Kuiter, R (2008) 'Fishes of Australia's southern coast ' (Reed New Holland Australia).

Gratwicke, B, Speight, MR (2005) The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. *Journal of Fish Biology* **66**, 650-667.

Guidetti, P (2000) Differences among fish assemblages associated with nearshore *Posidonia oceanica* seagrass beds, rocky–algal reefs and unvegetated sand habitats in the Adriatic Sea. *Estuarine, Coastal and Shelf Science* **50**, 515-529.

Guidetti, P (2007) Potential of marine reserves to cause community-wide changes beyond their boundaries. *Conservation Biology* **21**, 540-545.

Guidetti, P, Milazzo, M, Bussotti, S, Molinari, A, Murenu, M, Pais, A, Spanò, N, Balzano, R, Agardy, T, Boero, F, Carrada, G, Cattaneo-Vietti, R, Cau, A, Chemello, R, Greco, S, Manganaro, A, Notarbartolo di Sciara, G, Russo, GF, Tunesi, L (2008) Italian marine reserve effectiveness: does enforcement matter? *Biological Conservation* **141**, 699-709.

Halpern, BS (2003) The impact of marine reserves: do reserves work and does reserve size matter? *Ecological Applications* **13**, S117-S137.

Halpern, BS (2014) Conservation: making marine protected areas work. *Nature* **506**, 167-168.

Halpern, BS, Lester, SE, McLeod, KL (2010) Placing marine protected areas onto the ecosystem-based management seascape. *Proceedings of the National Academy of Sciences* **107**, 18312-18317.

Hardinge, J, Harvey, ES, Saunders, BJ, Newman, SJ (2013) A little bait goes a long way: the influence of bait quantity on a temperate fish assemblage sampled using stereo-BRUVs. *Journal of Experimental Marine Biology and Ecology* **449**, 250-260.

Harman, N, Harvey, ES, Kendrick, GA (2003) Differences in fish assemblages from different reef habitats at Hamelin Bay, south-western Australia. *Marine and Freshwater Research* **54**, 177-184.

Harrison, HB, Williamson, DH, Evans, RD, Almany, GR, Thorrold, SR, Russ, GR, Feldheim, KA, Van Herwerden, L, Planes, S, Srinivasan, M, Berumen, ML, Jones, GP (2012a) Larval export from marine reserves and the recruitment benefit for fish and fisheries. *Current Biology* **22**, 1023-1028.

Harrison, Hugo B, Williamson, David H, Evans, Richard D, Almany, Glenn R, Thorrold, Simon R, Russ, Garry R, Feldheim, Kevin A, van Herwerden, L, Planes, S, Srinivasan, M, Berumen, Michael L, Jones, Geoffrey P (2012b) Larval export from marine reserves and the recruitment benefit for fish and fisheries. *Current Biology* **22**, 1023-1028.

Harvey, ES, Cappo, M, Butler, JJ, Hall, N, Kendrick, GA (2007) Bait attraction affects the performance of remote underwater video stations in assessment of demersal fish community structure. *Marine Ecology Progress Series* **350**, 245-254.

Heck, KL, Hays, G, Orth, RJ (2003) Critical evaluation of the nursery role hypothesis for seagrass meadows. *Marine Ecology Progress Series* **253**, 123-136.

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

Holbrook, SJ, Kingsford, MJ, Schmitt, RJ, Stephens Jr, JS (1994) Spatial and temporal patterns in assemblages of temperate reef fish. *American Zoologist* **34**, 463-475.

Hutchins, B, Swainston, R (1986) 'Sea Fishes of Southern Australia.' (Swainston Publishing: Daglish, Western Australia).

Hyndes, GA, Platell, ME, Potter, IC, Lenanton, RCJ (1999) Does the composition of the demersal fish assemblages in temperate coastal waters change with depth and undergo consistent seasonal changes? *Marine Biology* **134**, 335-352.

Jenkins, CN, Van Houtan, KS (2016) Global and regional priorities for marine biodiversity protection. *Biological Conservation* **204**, 333-339.

Kelaher, BP, Coleman, MA, Broad, A, Rees, MJ, Jordan, A, Davis, AR (2014) Changes in fish assemblages following the establishment of a network of no-take marine reserves and partially-protected areas. *PLoS ONE* **9**, e85825.

Kelleher, G, Kenchington, R (1991) 'Guidelines for Establishing Marine Protected Areas.' (IUCN: Gland, Switzerland).

Kingsford, MJ (1989) Distribution patterns of planktivorous reef fish along the coast of northeastern New Zealand. *Marine Ecology Progress Series* **54**, 13-24.

Kirkman, H (2013) Choosing boundaries to marine protected areas and zoning the MPAs for restricted use and management. *Ocean and Coastal Management* **81**, 38-48.

Kirkman, H, Shepherd, SA (2015) Further efforts to protect biodiversity in coastal waters of South Australia. *Ocean and Coastal Management* **104**, 115-123.

Klages, J, Broad, A, Kelaher, BP, Davis, AR (2014) The influence of gummy sharks, *Mustelus antarcticus*, on observed fish assemblage structure. *Environmental Biology of Fishes* **97**, 215-222.

Langlois, TJ, Harvey, ES, Fitzpatrick, B, Meeuwig, JJ, Shedrawi, G, Watson, DL (2010) Cost-efficient sampling of fish assemblages: comparison of baited video stations and diver video transects. *Aquatic Biology* **9**, 155-168.

Lester, SE, Halpern, BS, Grorud-Colvert, K, Lubchenco, J, Ruttenberg, BI, Gaines, SD, Airamé, S, Warner, RR (2009) Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series* **384**, 33-46.

Lotze, HK, Lenihan, HS, Bourque, BJ, Bradbury, RH, Cooke, RG, Kay, MC, Kidwell, SM, Kirby, MX, Peterson, CH, Jackson, JBC (2006) Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* **312**, 1806-1809.

Lynch, TP (2006) Incorporation of recreational fishing effort into design of marine protected areas. *Conservation Biology* **20**, 1466-1476.

Malcolm, HA, Gladstone, W, Lindfield, S, Wraith, J, Lynch, TP (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**, 277-290.

Mora, C, Andréfouët, S, Costello, MJ, Kranenburg, C, Rollo, A, Veron, J, Gaston, KJ, Myers, RA (2006) Coral reefs and the global network of marine protected areas. *Science* **312**, 1750-1751.

Morrisey, DJ, Underwood, AJ, Howitt, L, Stark, JS (1992) Temporal variation in soft-sediment benthos. *Journal of Experimental Marine Biology and Ecology* **164**, 233-245.

Mosquera, I, Côté, IM, Jennings, S, Reynolds, JD (2000) Conservation benefits of marine reserves for fish populations. *Animal Conservation* **3**, 321-332.

Murphy, HM, Jenkins, GP (2010) Observational methods used in marine spatial monitoring of fishes and associated habitats: a review. *Marine and Freshwater Research* **61**, 236-252.

Nagelkerken, I, Grol, MGG, Mumby, PJ (2012) Effects of marine reserves versus nursery habitat availability on structure of reef fish communities. *PLoS ONE* **7**, e36906.

Palumbi, SR (2003) Population genetics, demographic connectivity, and the design of marine reserves. *Ecological Applications* **13**, S146-S158.

Parsons, DF, Suthers, IM, Cruz, DO, Smith, JA (2016) Effects of habitat on fish abundance and species composition on temperate rocky reefs. *Marine Ecology Progress Series* **561**, 155-171.

Pattengill-Semmens, CV, Semmens, BX (2003) Conservation and management applications of the reef volunteer fish monitoring program. (Eds BD Melzian, V Engle, M McAlister, S Sandhu, LK Eads.) pp. 43-50. (Springer Netherlands: Dordrecht).

Potter, IC, Claridge, PN, Hyndes, GA, Clarke, KR (1997) Seasonal, annual and regional variations in ichthyofaunal composition in the inner Severn Estuary and inner Bristol Channel. *Journal of the Marine Biological Association of the United Kingdom* **77**, 507-525.

Potter, IC, Loneragan, NR, Lenanton, RCJ, Chrystal, PJ, Grant, CJ (1983) Abundance, distribution and age structure offish populations in a western Australian estuary. *Journal of Zoology* **200**, 21-50.

Priede, IG, Merrett, NR (1998) The relationship between numbers of fish attracted to baited cameras and population density: studies on demersal grenadiers *Coryphaenoides (Nematonurus) armatus* in the abyssal NE Atlantic Ocean. *Fisheries Research* **36**, 133-137.

Roberts, CM, Hawkins, JP, Campaign, WES (2000) 'Fully-protected marine reserves: a guide.' (WWF Endangered Seas Campaign).

Rooker, JR, Dennis, GD (1991) Diel, lunar and seasonal changes in a mangrove fish assemblage off southwestern Puerto Rico. *Bulletin of Marine Science* **49**, 684-698.

Rosenberg, A, Bigford, TE, Leathery, S, Hill, RL, Bickers, K (2000) Ecosystem approaches to fishery management through essential fish habitat. *Bulletin of Marine Science* **66**, 535-542.

Russ, GR, Alcala, AC (2011) Enhanced biodiversity beyond marine reserve boundaries: the cup spillith over. *Ecological Applications* **21**, 241-250.

Russ, GR, Cheal, AJ, Dolman, AM, Emslie, MJ, Evans, RD, Miller, I, Sweatman, H, Williamson, DH (2008) Rapid increase in fish numbers follows creation of world's largest marine reserve network. *Current Biology* **18**, R514-R515.

Scholz, G, von Baumgarten, P, Wilson, H, Wright, A, Bryars, S, 2017. Monitoring, evaluation and reporting framework for marine parks program, DEWNR technical note 2017/06. Government of South Australia, Adelaide.

Smith, MPL, Bell, JD, Hair, CA (1991) Spatial variation in abundance of recently settled rocky reef fish in southeastern Australia: implications for detecting change. *Marine Ecology Progress Series* **77**, 95-103.

Soler, GA, Edgar, GJ, Thomson, RJ, Kininmonth, S, Campbel, SJ, Dawson, TP, Barrett, NS, Bernard, ATF, Galván, DE, Willis, TJ, Alexander, TJ, Stuart-Smith, RD (2015) Reef fishes at all trophic levels respond positively to effective marine protected areas. *PLoS ONE* **10**, e0140270.

Trevor, JW, Russell, BM, Russell, CB (2000) Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. *Marine Ecology Progress Series* **198**, 249-260.

Veale, L, Coulson, P, Hall, N, Hesp, A, Potter, IC (2015) Age and size compositions, habitats, growth and reproductive characteristics of a terapontid *(Pelates octolineatus)* in coastal waters. *Marine and Freshwater Research* **66**, 535-548.

Ward, TJ, Vanderklift, MA, Nicholls, AO, Kenchington, RA (1999) Selecting marine reserves using habitats and species assemblages as surrogates for biological diversity. *Ecological Applications* **9**, 691-698.

Warner, RR, Swearer, SE, Caselle, JE (2000) Larval accumulation and retention: implications for the design of marine reserves and essential habitat. *Bulletin of Marine Science* **66**, 821-830.

Watson, DL, Harvey, ES, Anderson, MJ, Kendrick, GA (2005) A comparison of temperate reef fish assemblages recorded by three underwater stereo-video techniques. *Marine Biology* **148**, 415-425.

Watson, DL, Harvey, ES, Fitzpatrick, BM, Langlois, TJ, Shedrawi, G (2010) Assessing reef fish assemblage structure: how do different stereo-video techniques compare? *Marine Biology* **157**, 1237-1250.

Whitmarsh, SK, Fairweather, PG, Huveneers, C (2017) What is Big BRUVver up to? Methods and uses of baited underwater video. *Reviews in Fish Biology and Fisheries* **27**, 53-73.

Willis, TJ, Anderson, MJ (2003) Structure of cryptic reef fish assemblages: relationships with habitat characteristics and predator density. *Marine Ecology Progress Series* **257**, 209-221.

Willis, TJ, Babcock, RC (2000) A baited underwater video system for the determination of relative density of carnivorous reef fish. *Marine and Freshwater Research* **51**, 755-763.

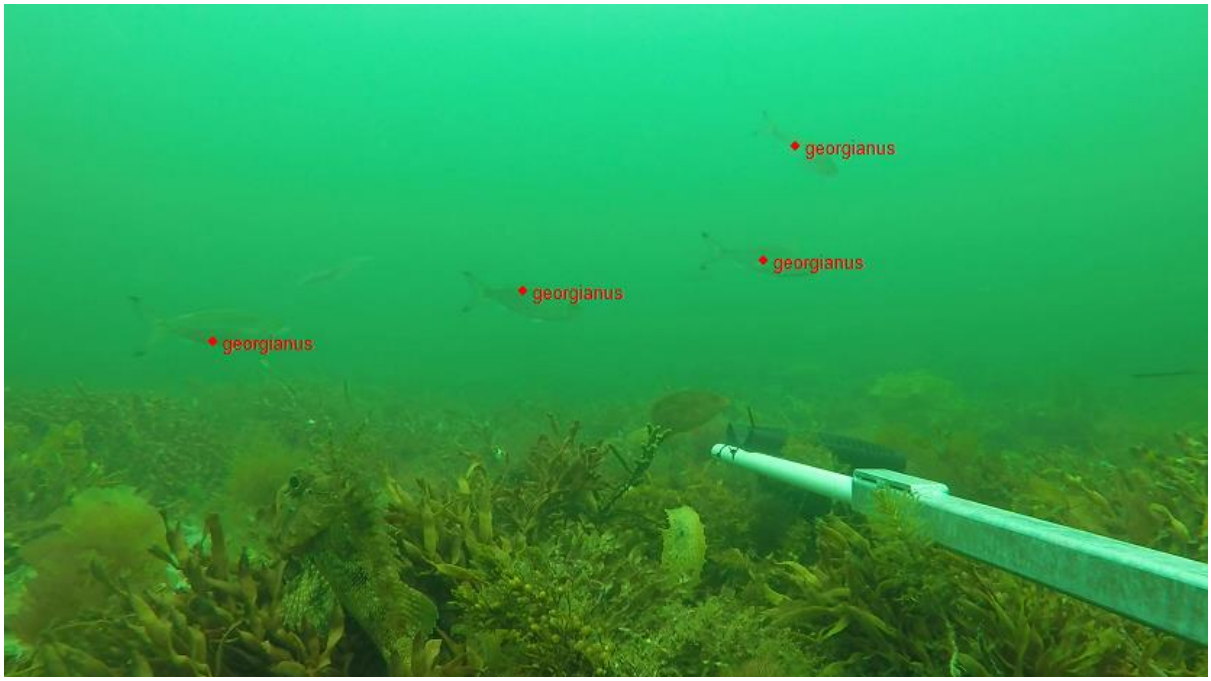
Willis, TJ, Badalamenti, F, Milazzo, M (2006) Diel variability in counts of reef fishes and its implications for monitoring. *Journal of Experimental Marine Biology and Ecology* **331**, 108-120.

Wright, JM (1988) Seasonal and spatial differences in the fish assemblage of the non-estuarine Sulaibikhat Bay, Kuwait. *Marine Biology* **100**, 13-20.

APPENDICES

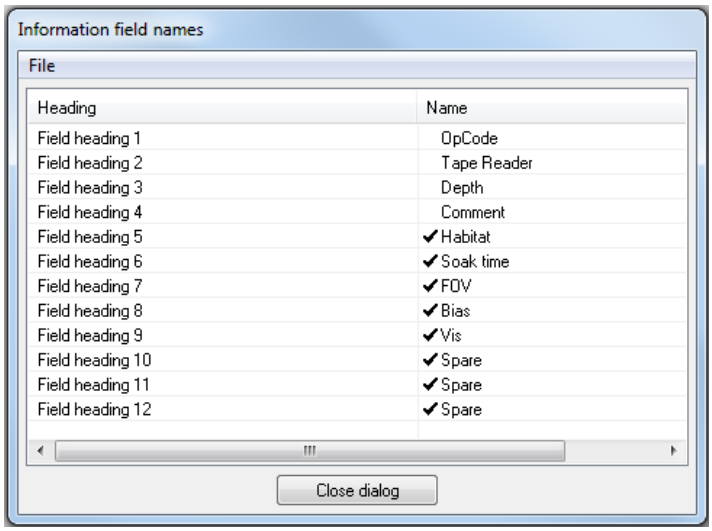
Appendix A. Example of *maxN* determination of *Arripis georgianus* from BRUV footage.

This is the maximum number of *A. georgianus* seen within the 60 minute BRUV footage. The fish are identified and once the footage has been watched and it is confirmed that this is the *maxN* of *A. georgianus,* each individual fish will be measured from this screen. This is completed in SEAGIS's EventMeasure.

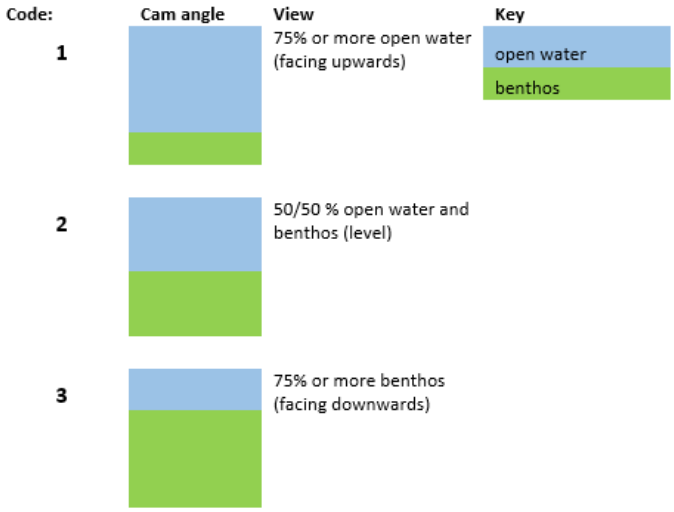


Appendix B. Standard parameters used on EventMeasure adapted from the Department of Environment, Water and Natural Resources

The information fields allow the variables between each drop site to be entered. The field names with a tick are fixed, and therefore cannot be changed. The remaining field headings can be changed to those seen below, and were used for the remainder of the project. The image view parameters are coded for in a numerical system based on the amount of bias, the field of view and the visibility of water.



Bias



Field of View

% view obstructed	Code
none	1
1/3	2
2/3	3
Full	4

Vis (visibility or water clarity)

Distance from cameras	Code
Cam to >4 m	1
Cam to 4 m	2
Cam to bait bag	3

Appendix C. Summary table of BRUV deployments, their locations and *maxN* and diversity.

	Sanctuary zone	Inside/outside	Number of deployments	Total *maxn*	Diversity (# of species)
Summer					
Coffin bay	1	I	2	61	6
Coffin bay	1	O	1	18	5
Coffin bay	3	I	2	48	3
Coffin bay	3	O	2	15	7
Coffin bay	4	I	0	-	-
Coffin bay	4	O	2	3	1
Venus bay	8	I	2	96	13
Venus bay	8	O	3	58	13
Winter					
Coffin bay	1	I	4	43	10
Coffin bay	1	O	4	37	14
Coffin bay	3	I	2	3	2
Coffin bay	3	O	2	1	1
Coffin bay	4	I	1	0	0
Coffin bay	4	O	2	13	5
Coffin bay	5	I	2	18	1
Coffin bay	5	O	2	16	3
Venus bay	8	I	4	36	6
Venus bay	8	O	2	9	2

Appendix D. Pairwise comparisons between depths, habitats and sanctuary zone locations.

Pairwise comparisons between depths. Significant differences (P<0.05) are in bold. Due to multiple testing we would expect 1 in 20 tests to be significant by chance alone.

SITE: DEPTHS	T	P
2m, 1m	0.7391	0.7210
2m, 4m	1.4969	0.0638
2m, 6m	1.9360	**0.0044**
2m, 3m	1.5466	**0.0430**
2m, 5m	1.9956	**0.0033**
1m, 4m	1.4158	0.0662
1m, 6m	1.7362	**0.0070**
1m, 3m	1.4925	0.0550
1m, 5m	1.7667	**0.0116**
4m, 6m	1.0812	0.2566
4m, 3m	1.1680	0.2045
4m, 5m	1.2768	0.2243
6m, 3m	1.3713	0.2025
6m, 5m	2.0472	**0.0270**
3m, 5m	1.7098	0.1317

Pairwise comparisons between habitats. Significant differences (P<0.05) are in bold.

SITE: HABITAT	T	P
Seagrass, broken sand	1.7255	**0.0108**
Seagrass, seaweed	1.2965	0.1212
Seagrass, sand	1.4319	0.0675
Seagrass, sponge	1.4014	0.0794
Broken sand, seaweed	1.1280	0.3787
Broken sand, sand	1.0990	0.3043
Broken sand, sponge	0.8961	0.7537
Seaweed, sand	1.2001	0.2876
Seaweed, sponge	No test	No test
Sand, sponge	1.0747	0.4349

Pairwise comparisons between sanctuary zone locations. Significant differences (P<0.05) are in bold.

SITE: SANCTUARY ZONE LOCATION	T	P
SZ 8, SZ 3	1.4094	0.1064
SZ 8, SZ 1	2.1426	**0.0005**
SZ 8, SZ 4	2.4887	**0.0047**
SZ 8, SZ 5	2.0154	**0.0059**
SZ 3, SZ 1	2.2400	**0.0003**
SZ 3, SZ 4	1.7947	**0.0203**
SZ 3, SZ 5	2.0239	**0.0086**
SZ 1, SZ 4	2.5656	**0.0006**
SZ 1, SZ 5	2.0142	**0.0123**
SZ 4, SZ 5	3.0509	**0.0082**