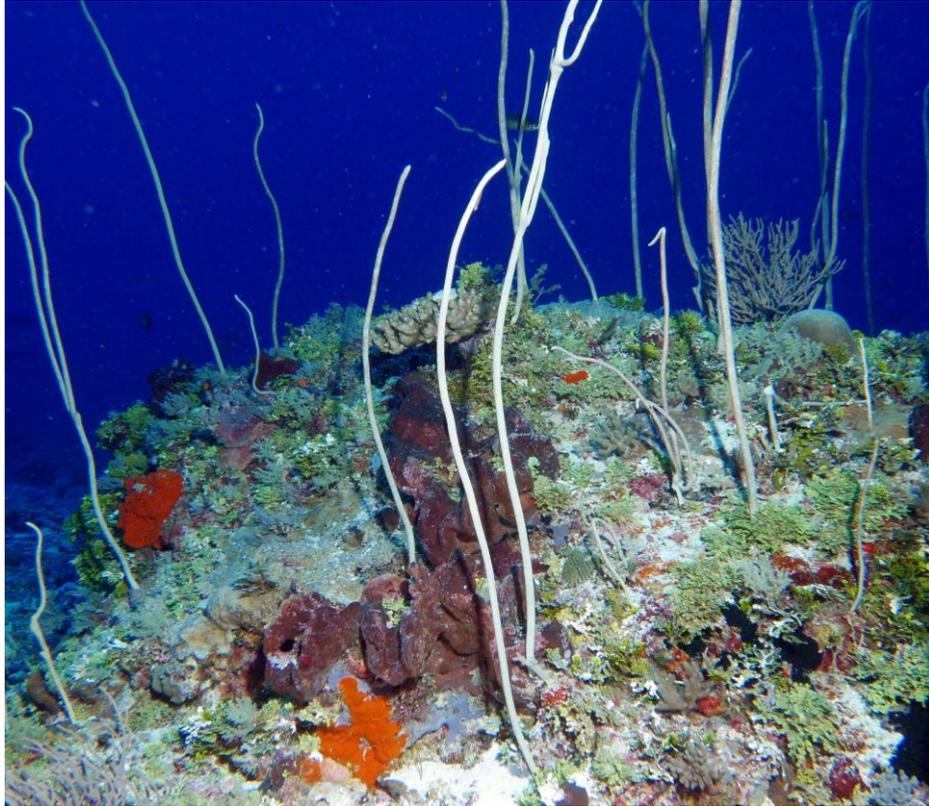# The Barracouta, Goeree and Vulcan, Shoals Survey 2016

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Revision history

Issue	Date	Originator	Reviewed	Approved
A	01/09/2017	Ben Radford	Andrew Heyward	Michaela Dommisse

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Executive Summary

Completion of a 2016 seabed biodiversity survey on the three submerged shoals, Barracouta East, Goeree, and Vulcan provided an updated appraisal of their status and follows previous surveys in 2010, 2011 and 2013. These three shoals were part of the original set of shoals studied in relation to the Montara uncontrolled hydrocarbon release, which occurred in 2009. Two of the shoals, Vulcan and Barracouta East were originally sampled with towed cameras, as a pilot study in 2010. All three shoals were a subset of the group of nine shoals that were surveyed in more detail in 2011 and located at varying distances and potential exposures from the Montara Well Head Platform (MWHP).

Prior to the Montara hydrocarbon release, there were no baseline data for benthic habitats and associated fish communities on these shoals. The major surveys in 2011 documented these seabed features as diverse and sensitive ecosystems. Changes in species richness and abundance were observed on shoals between 2010 and 2011, as well as between individual shoals more generally, but it was unclear as to what degree these differences reflected natural year to year variability. Barracouta East, Goeree, and Vulcan were subsequently revisited in 2013 to gather further information on how the shoal seabed biota might vary annually. Using the same sampling field methods applied during the initial post-Montara shoal assessments, the most recent survey in 2016 has extended the available data, allowing insights into temporal variability across years for key aspects of the benthic habitats and their associated fish communities. These ecological research represent the unique time series record for shoal features.

The aim of the present study was to characterize multi-year changes in benthic and fish assemblages between successive surveys in 2010, 2011, 2013 and 2016 on these three shoals. In order to facilitate comparisons, the same techniques and key deployment sites used in the 2011 surveys on each shoal were also targeted in the 2013 and 2016 studies. Shoals were surveyed using towed video and still camera transects for benthos and stereo Baited Remote Video Systems (BRUVS™®) for fish communities.

A variety of algae were the most abundant benthic group encountered on all shoals. Overall mean coral cover differed between individual shoals, within a range of 5-15%, but varied from survey to survey. Both declines and increases in major benthic categories, such as hard corals, other primary producers and filter feeders were recorded between successive surveys at all three shoals. The observed temporal variability did not have a consistent trajectory and was unrelated to distance from the MWHP and instead the result of ecological processes (recruitment, mortality, turn over and effects of disturbance like storms). This time series study of benthic cover on shoal features is the first of its kind, its important baseline research with which to understand the patterns of natural turnover and perturbation in coming years.

In addition to extending the information base to 2016, the multiyear data set also provided an opportunity to test and evolve new analytical methods for future rapid reporting of shoal status. Using the image-derived benthic data obtained during the first three surveys, a computer-based habitat classification system was evaluated and validated against manual classification by marine biologists. An artificial intelligence (AI) classification system developed at AIMS, named BenthoBot, demonstrated a robust classification performance for major benthic categories such as hard and soft corals.

The AI system can process images in the order of 100 times faster than trained technicians and did not suffer the same level of inter-observer variation introduced by different biologists. Consequently AI-based image analysis by BenthoBot type systems are likely to play a part in future rapid and cost effective monitoring of seabed habitats.

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A total of 362 fish species have been recorded at the three shoals in 2011, 2013 and 2016s, with 137-165 species recorded on individual schools during individual surveys. The observed fish communities have remained largely stable in terms of species richness and abundance at all three shoals. Overall diversity and abundance of fish did vary, both between shoals and by year, but the greater variability in fish communities was associated with "shoal" rather than with "year".

Fish were more diverse and abundant at the shallower sites <30 m. Species richness was consistently higher at Barracouta East Shoal, however, fish abundance remained relatively consistent, with no clear trend observed among shoals or years.

Conclusions and suggestions for the future

This information extends understanding of biodiversity patterns on these shoal features in space and time, with the aim of better understanding natural variability versus possible anthropogenic effects. It is clear that the shoals support diverse benthic communities, with the plateau regions receiving enough light at the seabed to support benthic primary producers and many species typical of tropical coral reef ecosystems. Major habitat types on the shoal plateaus have persisted over the period 2010-2016, but the abundance of key biota has varied in time.

The additional surveys in 2013 and 2016 have revealed variations in the abundance and diversity of both the benthic assemblages and the associated fish communities, with the temporal trends reflecting disturbance and recovery at the individual shoal level, rather than suggesting any clear correlation with potential exposure to the uncontrolled release of 2009.

Future monitoring of the benthos, is likely to become increasingly cost effective with the ongoing development of automation for both field data acquisition and subsequent analysis. A comparative study of these shoals features with similar features along the Great Barrier Reef and Pilbara coastlines indicate these shoals should be regarded as biodiversity hotspots for both benthic and fish communities. However, as the effects of hard coral cover, depth and shoal appear to structure the fish communities to a greater degree than changes observed through time, benthic monitoring may provide an effective guide to the overall status of the biological communities present.

1. Introduction & Background

Between the 21st August and the 3rd November 2009, loss of control from the Montara well resulted in the release of gas and crude oil from the Montara reservoir through the Montara Well Head platform (MWHP) into the Timor Sea. On the 9th October 2009, the lease operator PTTEP Australasia (Ashmore Cartier) Pty. Ltd (hereafter PTTEPAA) and the Department of Sustainability, Environment, Water, Population and Communities, DSEWPaC) released a Monitoring Plan for the Montara Well Release Timor Sea (PTTEPAA 2009).

In 2010, PTTEPAA requested a preliminary assessment of benthos on the tops of Barracouta and Vulcan Shoals, two submerged shoals within 100 km of the MWHP that were likely to intersect the hydrocarbon plume. The purpose of the survey was to provide preliminary data to aid in planning for a broader assessment of the numerous shoals in the region, should it be required as a result of a triggering of the benthic program (component S)5 of the Montara Monitoring Plan (PTTEPAA 2009).The benthic communities of these shoals were surveyed using towed video to provide an initial set of benthic habitat data (Heyward et al. 2010).

Triggering of the S5 component of the Monitoring Plan initiated a larger study of shoals in 2011. The 2011 survey extended field sampling to include repeat surveys of Vulcan and Barracouta Shoals and initial surveys of a further seven banks and shoals, using previously adopted methods of towed video and still photography, as well as multibeam acoustic swath mapping to characterise habitats, and BRUVS™ to describe fish communities. That research project was carried out to quantify physical characteristics, identify and characterise the benthic and fish communities, estimate the potential exposure to surface oil and dispersed oil, and identify any obvious damage to the associated communities (Heyward et al, 2012).

Subsequent surveys of three example shoals, Vulcan, Barracouta East, and Goeree, have been designed to gain insights into how benthic and fish communities might have changed since 2011. Goeree and Vulcan Shoal are the two shoals closest to the MWHP, with Barracouta East Shoal significantly further away. It was envisaged that these follow up surveys would an assessment of other benthic habitats through time, and among shoals. As Goeree Shoal was surveyed in 2011, 2013 and 2016, while Vulcan and Barracouta East Shoal comparisons could also include the 2010 survey.

The 2013 report characterised multi-year changes in benthic assemblages between successive surveys on these three shoals as:

- marked increase of macroalgae, along with sand, and unconsolidated substrate (rubble) at all shoals (with some and significant variation between shoals and time periods)
- significant declines in the visual abundance of consolidated reef, along with most hard coral, soft coral, and sponge categories
- complete loss of seagrass on Vulcan Shoal (following on from the initial major decline between 2010 and 2011 surveys)
- loss of a soft coral habitat (predominantly a mono-specific community of *Nephthea* sp.) that was present on the western side of Barracouta East Shoal in both 2010 and 2011 surveys

The September 2016 final survey was undertaken at the same three shoals, repeating methods and sampling locations used in the 2013 survey. This information extends our documentation of

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biodiversity patterns at these shoals over a 5-6 year period, broadening our understanding of the dynamics and variability in community structure shown within and between shoals through time.

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2. Banks & Shoals Habitat Mapping

2.1 Introduction/Background on 2011 observations

The shoals have been surveyed three times previously and this report provides data about the benthic community from the most recent survey in September 2016, with data from April 2010, March 2011 and April 2013 included for comparison. All primary images and all point-intercept derived data from the Project, produced by biologists classifying photos from surveys between 2010-2016 have been archived at AIMS. In this report computer-based image analysis has been applied in order to provide a consistent and robust synthesis of temporal trends over the six year span of this project. An artificial intelligence engine called BenthoBot was used to re-analyse all seabed images from all years 2010-2016, processing each image using exactly the same approach.

2.2 Methods

The location of the three shoals in relation to the MWHP can be seen in Figure 1. The tops of the shoals down to a depth of about 60 m were surveyed using a towed video system and followed protocols described in the previous report (Heyward et al. 2013). The towed video system captures forward projected video of the community at a height of about 1 m from the seafloor, and downward facing still images of the benthos using a high resolution camera. The benthic community was characterised by analysing the still camera images collected on established transects across the shoals.

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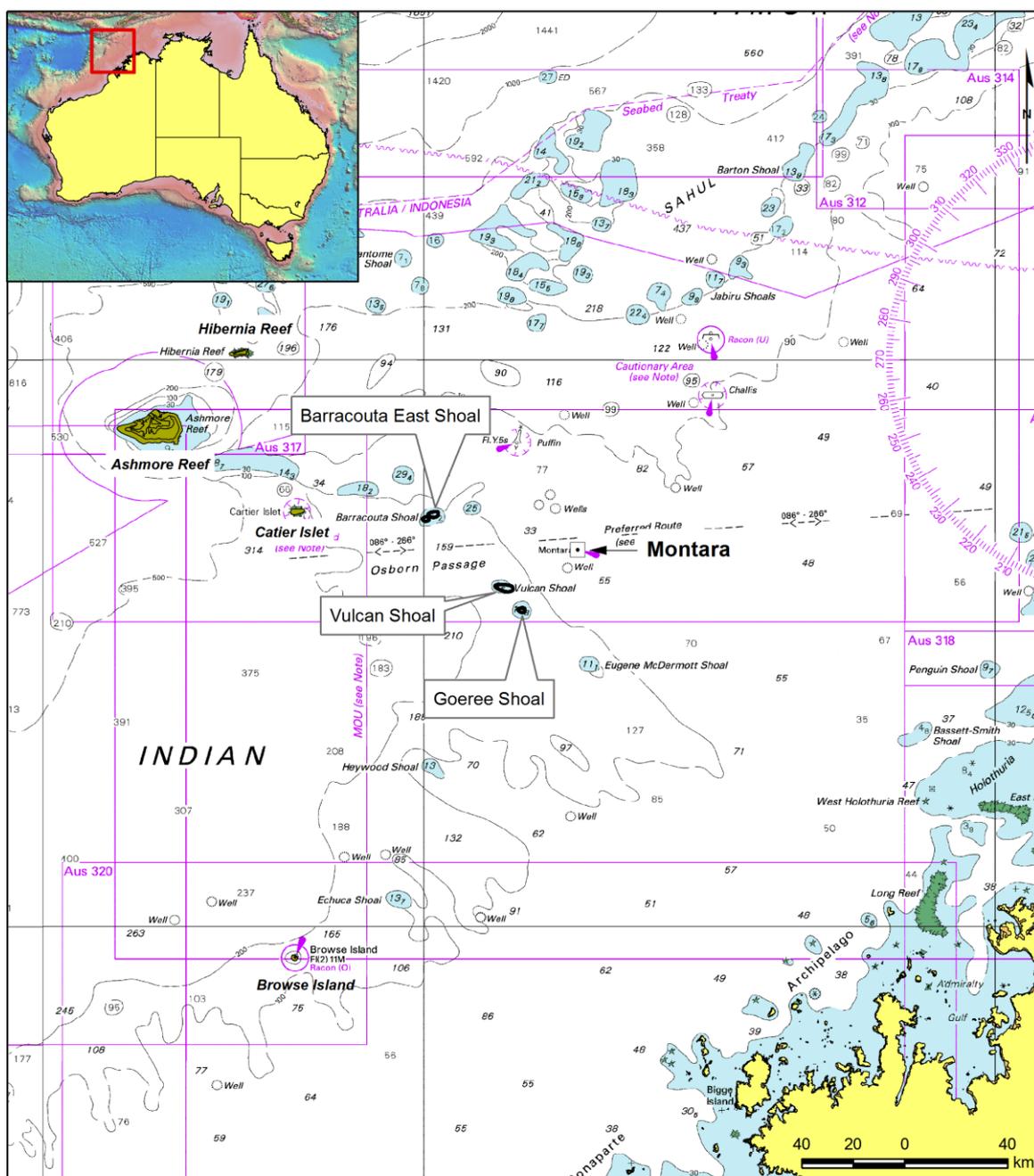
Figure 1. Location of Barracouta East, Goeree and Vulcan Shoal study sites surveyed between 2010 -2016 in relation to the location of the Montara well head.

2.2.1 Shoals, transect length and position

In 2010, only Barracouta East (Figure 2) and Vulcan Shoals (Figure 3) were surveyed, using 500 m long transects arranged randomly on the shoals' plateau regions. In 2011, individual transect length was increased on all shoals to extend sample coverage and at the same time increase survey efficiency by requiring fewer deployment-recovery episodes of the towed camera gear. Goeree Shoal (Figure 4) was included in the 2011 survey and adjustments were made to transect length (increased to approximately 1.42 km) and position (arranged grid-like to increase shoal-wide coverage), largely to improve the capacity of the design to support spatial modelling with the data and enable building of benthic habitat maps. The 2011 survey design was retained for subsequent surveys in 2013 and 2016,

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and transects were repeated by returning to each transects start waypoints and using the ship's navigation to head towards the transect end waypoints (see Figures 2, 3 & 4; strong currents can reduce exact transect alignment).
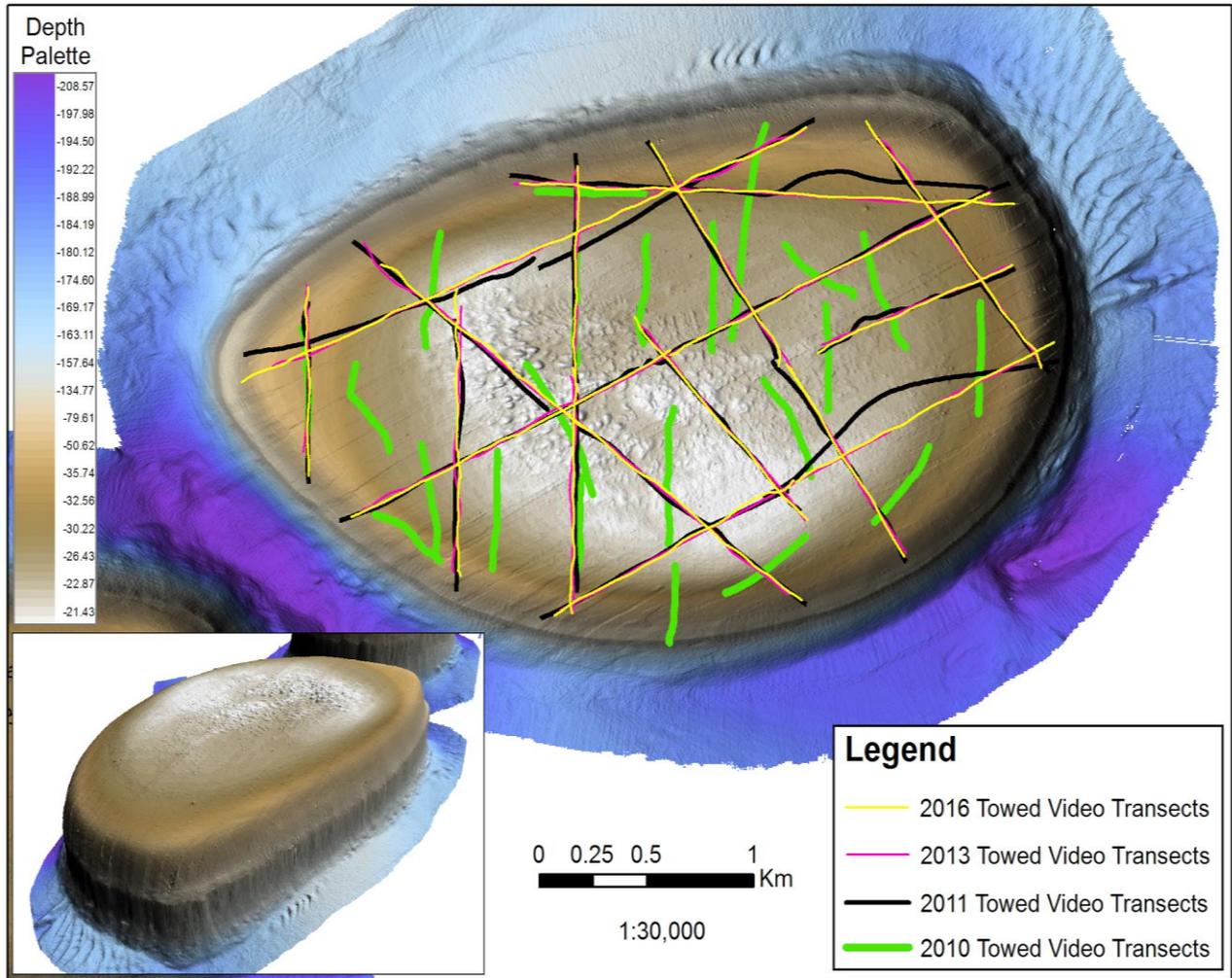
Figure 2. Map of towed video transects at Barracouta East Shoal for the four surveys.
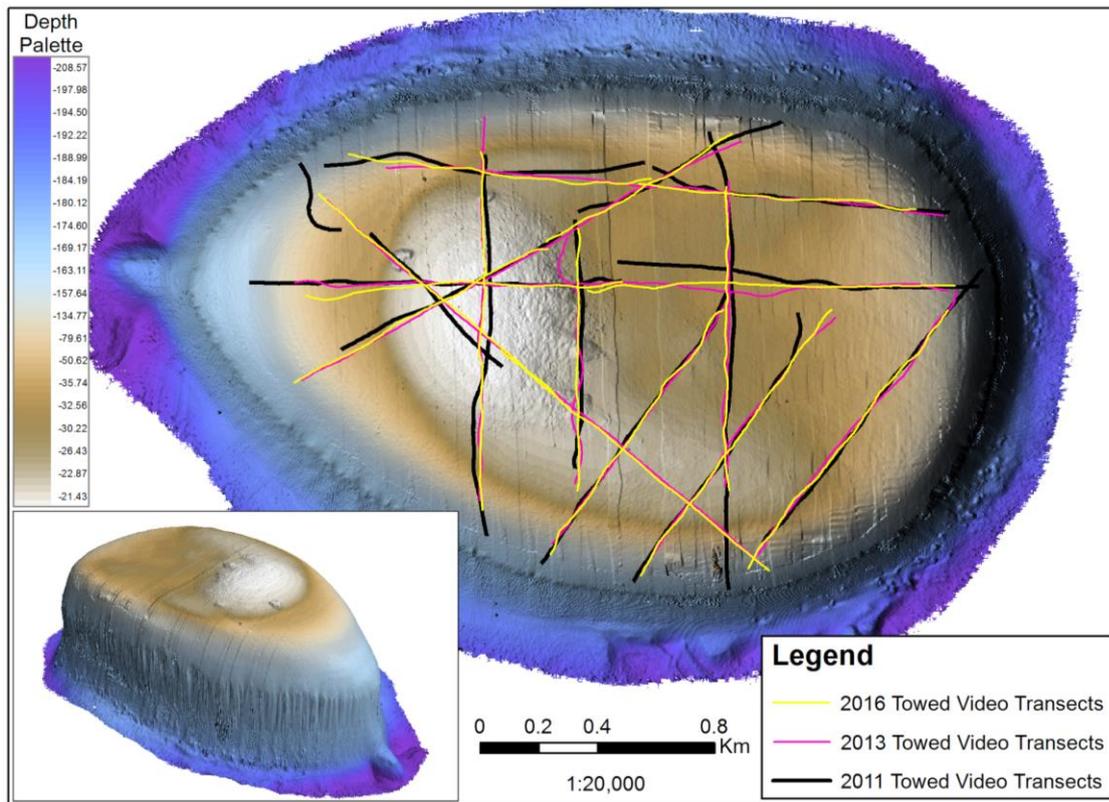
Figure 3. Map of towed video transects at Goeree Shoal for the three surveys (Goeree Shoal was not surveyed in 2010).
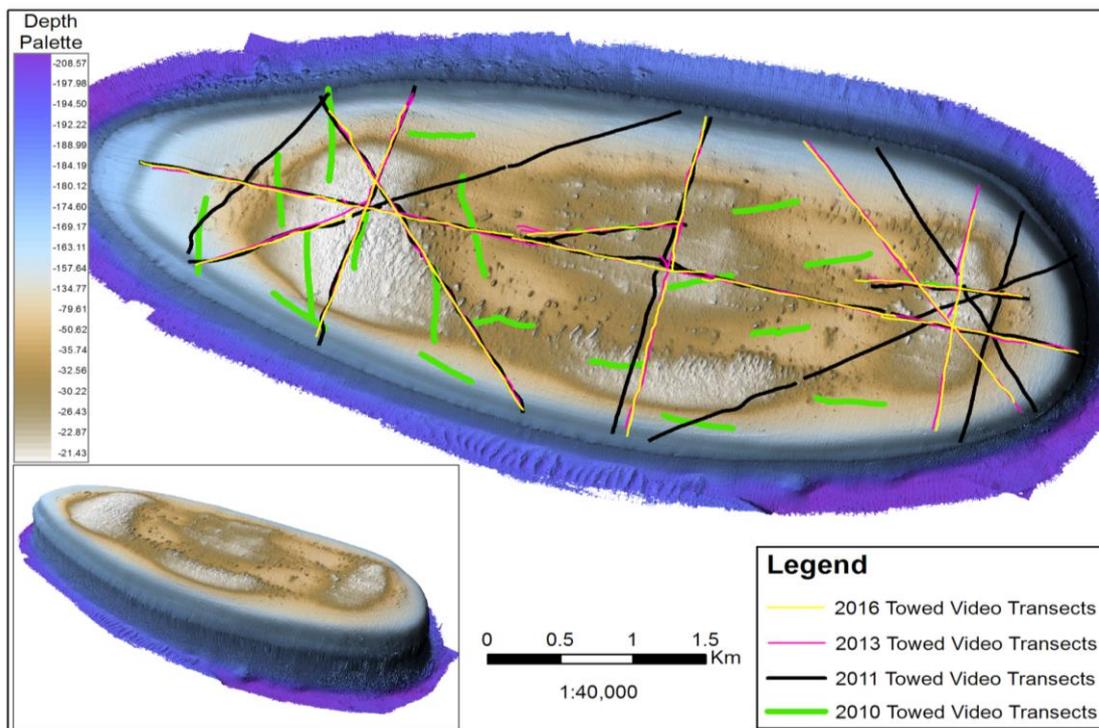
Figure 4. Map of towed video transects at Vulcan Shoal for the four surveys. Two transects on the eastern end were repositioned to better cover habitats. Three transects were not surveyed in 2013 and 2016 due to time constraints.

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2.2.2 Still Image Analysis

Data on benthic composition was extracted from high resolution images collected at regular intervals along each transect. Image spacing varied from 6-10 m (based on previous experience of benthic habitats in the bioregion (Heyward *et al.* 1997, Heyward and Rees 1999, Heyward *et al.* 2010) and the number of images collected and used in the analysis was proportional to the size of the shoal and total length of the transect conducted.

Considerations for Point sampling of images: In all survey years still images were analysed by marine science technicians using a point intercept method (see Heyward et al, 2012). Optimizing the image analysis protocols required trade-offs between the level of detail extracted from each photo and the time required for laboratory image analysis by marine biologists. Analysis of 2010 high resolution benthic photos along the towed video transects at Vulcan (19 transects) and Barracouta (20 transects) Shoals, comparing the use of 5, 10, 15 and 20 points per photo, indicated that major patterns of biotic distribution and transect proportion abundance are very similar represented by these various sampling intensities. There are some subtle differences between how some biotic groups are represented, however, to what extent these differences are interpretable is unknown. More details of analysis outcomes are summarised below. The total number of biotic groups detected using 20 points detected 7 more biotic groups compared the lowest sampling intensity of 5 points. However, in all cases, these additional groups found in the 20 point interrogation of images were at trace levels and that the proportional relationships of biota along transects was overall very similar between varying point sampling intensities. All point sampling intensities produces robust patterns with a range of multivariate analysis. Subsequently the 5 point sampling protocol was adopted and applied again in 2013 and 2016.

Manual classification remains a rate limiting process, even when applied to 5 points per photo. The 2016 transect imagery consisted of 7,642 high resolution photos, which might require an average of 75 person days to classify. Processing speed is dependent on the technician's experience, familiarity with the biota and level of biological complexity in the images. An additional consideration is variation in classification between multiple technicians. Some artefacts invariably arise due to alternate decisions amongst technicians about identification of some taxonomic groups. These differences can lead to spatial artefacts when image analysis is distributed among multiple technicians to process a single survey. Additionally, when surveys from different years are compared temporal mismatches can be perpetuated if inter-observer classification is not fully calibrated. Alternatively "adaptive learning" by technicians may improve identifications from year to year, but generally older datasets are not reanalysed.

All primary images and all point-intercept derived data, from surveys between 2010 -2016 have been archived. However, for this report an alternative approach to image analysis has been applied in order to provide a consistent and robust synthesis of temporal trends over the six year span of this project. An artificial intelligence engine called BenthoBot was used to re-analyse all seabed images from all years 2010-2016, processing each image using exactly the same approach. BenthoBot is a computer algorithm developed to classify points on an image, based on the spectral properties extracted from each image. It has been developed specifically by the Australian Institute of Marine Science to provide an efficient and consistent means of generating the point based broad scale benthic classification data that underpins both our offshore shoals and shallow reef based long term monitoring programs. The benefits of using BenthoBot include standardisation of the number of points sampled per image across all years (20 points per image) and removal of inconsistency in point classification associated with numerous technicians scoring images.

2.2.3 BenthoBot: model development, validation and testing

BenthoBot uses a state of the art 50 layer residual neural network - a convolutional neural network designed by Microsoft research (He *et al.* 2015). The images for all years of the study were first corrected using contrast stretching. For every point on an image classified by a technician, a 256 x 256 pixel patch was cut around the point. The patches were then randomized and separated into three sets of data: 60% training ~10,000 patches; 20% validation ~3700 patches; 20% test ~3700 patches. The number of patches per benthic category were capped at 1500 to prevent introducing bias into the model for a category. The model was developed using the training and validation data, and the model's performance was validated on the test data.

15 benthic categories were used to train the BenthoBot algorithm. These categories were based on the 23 categories used in the 2013 report, however BenthoBot was not able to classify to the same fine-scale. Loss of resolution was highest for hard coral which went from nine fine-scale groups used by technicians (*Acropora* branching, *Acropora* digitate/corymbose; *Acropora* tabulate; Branching; Encrusting; Foliose; Massive; Submassive/columnar; Free-living) to three groups discernible by BenthoBot (branching, non-branching and free-living). Sponge reduced from three groups (encrusting; erect/branching; massive) to two groups (erect/branching; non erect/branching) and soft coral from two groups (soft coral; gorgonians) to one group (soft coral).

The confusion matrix (Figure 5) shows the Recall value, which is the accuracy of the prediction made by the model on the test data i.e. the predicted category against the true category and identifies the most likely groups each could be "confused" with. Groups that resolved well include hard coral (branching 0.76 and non-branching 0.62), seagrass (0.74), soft coral (0.74), sand (0.67), *Halimeda* (0.65) and other organisms (0.61). Groups that were poorly resolved, such as sponges and algae, contained a high proportion of encrusting growth forms (excluding *Halimeda*). Encrusting organisms, and turf algae often display similar colour spectrums and minimal structural complexity which makes identification difficult for both technicians and computer algorithms. Consolidated (0.23) and Unconsolidated groups (0.33) have encrusting and turfing organisms covering them in varying degrees and this continuum across biotic and substrate types translates to poor resolution among these groups (also true for technician classifications). Sponges with erect/branching growth form were expected to have good detection but did not rate well in the matrix (0.13) due to low presence at this location and consequently a low number of training points. Consequently the two sponge categories were grouped for the study analyses. Additional measures summarising the performance of BenthoBot on the test data are in Table 1.
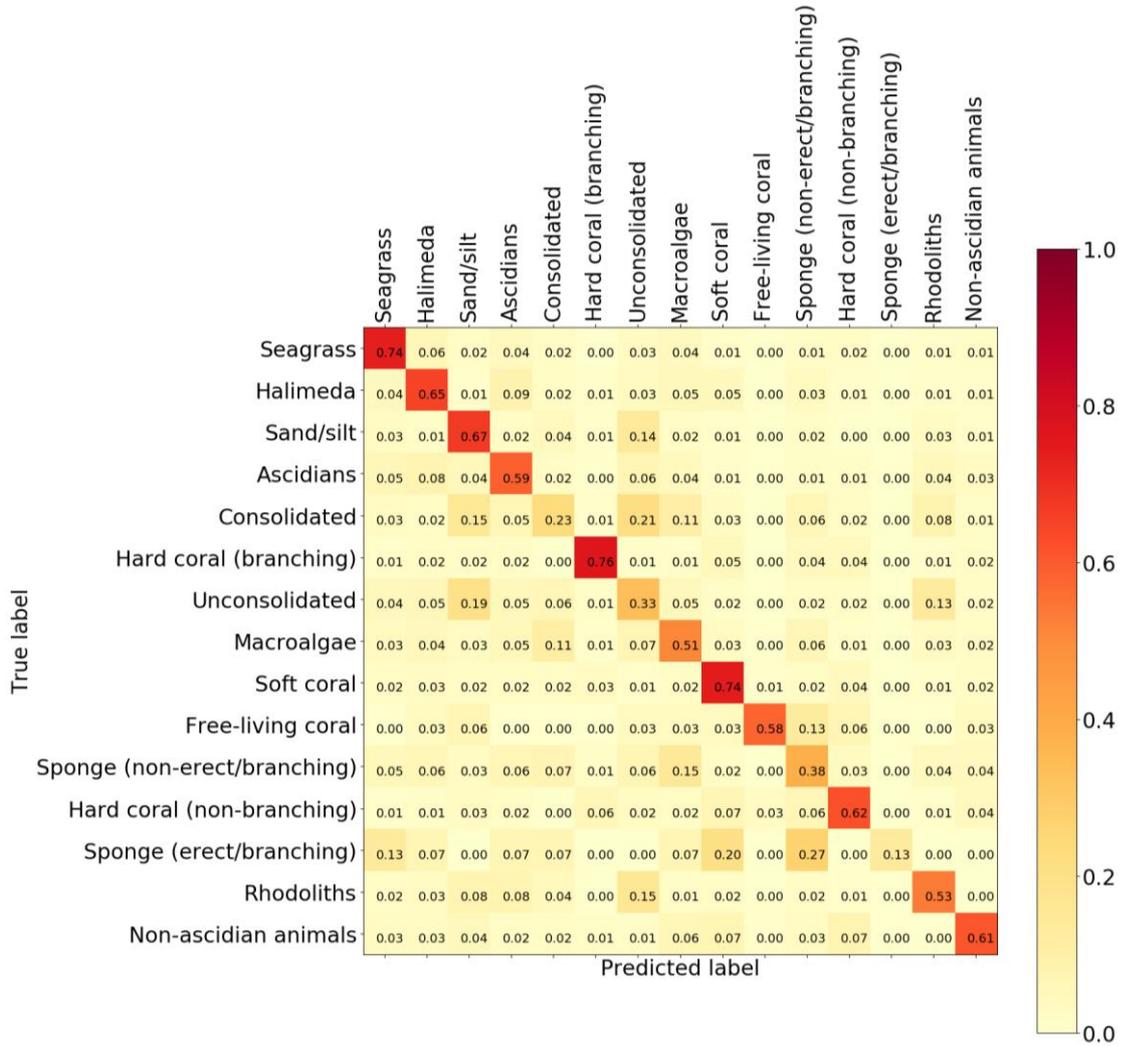
Figure 5. Confusion matrix showing the accuracy of BenthoBot predicted labels against the true label across 15 benthic groups.

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Table 1. Precision, Recall and F1 scores for each of the benthic categories based on the test data validation. *Precision* measures relevancy i.e. the accuracy of the prediction made by the model. *Recall* measures how many relevant results were returned i.e. how much of the test data was positively predicted for a particular category. *F1-score* is a weighted average of precision and recall. *Support* is the number of points/patches used in the testing dataset for a particular benthic category. The highest performing five categories are listed in red. For final categories used see table 2

Benthic category	Precision	Recall	f1-score	Support
Hard coral (branching)	0.76	0.76	0.76	197
Soft coral	0.74	0.69	0.71	312
Seagrass	0.74	0.66	0.69	280
Hard coral (non-branching)	0.62	0.71	0.66	295
Non-ascidian animals	0.61	0.67	0.64	229
Halimeda	0.65	0.59	0.62	282
Sand/silt	0.67	0.52	0.59	326
Free-living coral	0.58	0.55	0.56	31
Ascidians	0.59	0.51	0.55	269
Rhodoliths	0.53	0.54	0.54	269
Macroalgae	0.51	0.48	0.49	301
Sponge (non-erect/branching)	0.38	0.5	0.44	302
Unconsolidated	0.33	0.31	0.32	323
Consolidated	0.23	0.36	0.28	310
Sponge (erect/branching)	0.13	1	0.24	15
avg / total	**0.56**	**0.55**	**0.55**	**3741**

2.2.4 BenthoBot classification of the survey images (2010, 2011, 2013 and 2016)

BenthoBot, using the trained model developed for our dataset, was run on the four years of survey images to produce a new dataset of classifications. 20 points (patches) per image were classified which was substantially higher than the five points previously classified by technicians. Classifying a greater number of points improves detection of rarer categories (seen as fine slithers on pie charts) and potentially offsets loss in precision associated with uncertainty in the BenthoBot procedure. For each BenthoBot-classified point the algorithm returns a vector of the probability that point belongs to one of the 15 benthic categories. This vector of probabilities was converted to a categorical score for each point via a call to the rmultinom function in R. This categorical point score data exactly replicates the data typically generated by manual classification by technicians, and was treated as such in all subsequent statistical analyses.

2.2.5 Benthic categories

Changes in the broad-scale benthic composition among the four sample years and three shoals were examined using the same nine major categories as the previous report (see Table 2, first column). We focused a finer-scale benthic analysis on five sub-categories representing important ecosystem indicators, that were also shown to have an f1-score of greater than 0.60 (highlighted in red, Table 2), including: hard coral (non-branching), hard coral (branching), soft coral, seagrass and *Halimeda*. In addition, we examined two grouped categories 1) filter feeders and 2) primary producers (see Table 2 for details on biota included in these groups) that represent high level indicators of broad scale ecosystem change and captured benthic groups not always well resolved by BenthoBot (such as sponge and algae). Non-ascidian animals had an f1-score of 0.64 but were not included in the analysis as this group included a range of unrelated biota (many motile).

Table 2. Major and sub-categories scored by BenthoBot showing F1-scores (a measure of precision and recall) and percentage cover across the entire study. The 2 Sponge categories were combined. 5 sub-categories included in the finer-scale analyses are highlighted in red. Some categories were Grouped together (highlighted in blue) to improve the f1-score and inclusion in the analysis.

Major category	Sub-category	Description	f1-score	% cover
1.Hard coral	1. HC (non-branching)	Scleractinian hard corals with non-branching growth form (i.e. encrusting, foliose, submassive/columnar, massive, corymbose, digitate, tabulate)	0.66	5.3
	2. HC (branching)	Scleractinian hard corals with branching growth form	0.76	2.7
	3. Free-living corals	Hard corals that are not attached to the sea floor; *Ctenactis, Cynarina, Diaseris, Fungia, Halomitra, Herpolitha, Polyphyllia, Scolymia*	0.56	0.7
2. Algae	4. Macroalgae	Any algae described as macroalgae; excludes *Halimeda,* turf algae	0.49	10.3
	5. *Halimeda*	*Halimeda* spp.	0.62	4.5
	6. Rhodoliths	Rhodoliths	0.54	7.8
3. Other organisms	7. Non-ascidian animals	Non-ascidian animals, including: bryozoans, hydroids, urchins, zoanthids, anemones, annelids, crinoids, holothurians, corallimorphs, starfish, gastropods, *Millepora*	0.64	4.1
	8. Ascidians	Represented overwhelmingly by the small photosynthetic Ascidian *Lissoclinum* sp. (see Heyward et al. 2010)	0.55	3.4
4. Sponge	9. Sponge	All sponge growth forms	0.34	5.2
5. Soft coral	10. Soft coral	All soft corals (includes Gorgonians)	0.71	4.9
6. Seagrass	11. Seagrass	All seagrasses	0.69	3.7
7. Sand/silt	12. Sand/silt	Sand and silt, coarse sand	0.59	26.3
8. Consolidated	13. Consolidated	Consolidated substrate; reefal substrate, turf and crustose coralline algal	0.28	6.4
9. Unconsolidated	14. Unconsolidated	Rubble, rocks, shells/skeletal rubble, stones	0.32	14.7

Grouped categories	Description	f1-score	% cover
1. Primary producers	Algae (all), seagrass	0.68	26.2
2. Filter feeders	Hard coral (all), sponge, soft coral, ascidians	0.76	22.3

2.2.6 Data management and security

All data were collated and archived at AIMS. Derived data files were added to the PTTEP archive. Position and depth data derived from the ship's navigation package were associated with all field sampling. The PTTEP towed video data and associated digital stills have been georeferenced and stored in our Oracle Towed Stills database. All raw video and images, raw field data, field notes and auxiliary files were archived on our local AIMS WA server. Towed video position, depth and habitat classification data were transferred in Microsoft Access database structure and all video was archived in avi format. Still photos associated with each towed video transect were recorded in jpeg format and georeferenced. Metadata for all sampling is included in Appendix 2 and MEST Metadata records and locational ESRI Shape files have been created and published to our internal geoserver which can be viewed through the AIMS intranet.

The AIMS WA server mirrors onto Townsville servers for archival and data protection purposes. All data held on AIMS servers use weekly/monthly normal tape backup routines. AIMS IT manages all internal user accounts and permissions. The AIMS WA Data Manager controls access to the data storage and spatial index for data collected by AIMS WA staff.

2.3   Data Analysis

Images less than 10 m depth and greater than 50 m depth were excluded from the analyses to ensure representation of depths across shoals and years remained consistent (see Figure 6). In 2010 (pilot study) transect length was shorter (reflected by the low number of images) and Goeree shoal was not surveyed (Figure 6, dark grey). To investigate the differences in community structure associated with depth, the data were divided into two depth bands (shallow: 10 m-30 m; deep >30 m - 50 m) with 30 m representing the upper depth for mesophotic coral ecosystems. Goeree Shoal has a deeper plateau than the other two shoals and most of the images were below 30 m.
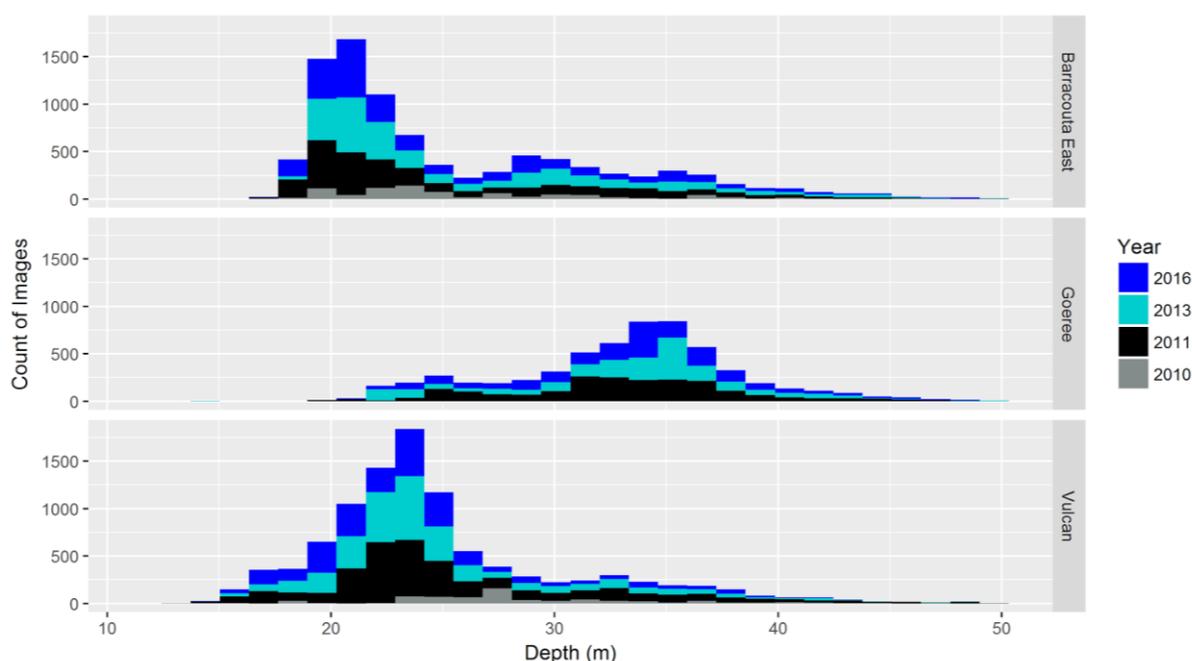
Figure 6. Stacked bar graphs showing the distribution of images by shoal, year and depth. The total value of the bar is all the year values added together. The plateau at Goeree Shoal was deeper than Barracouta East and Vulcan Shoal.

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Percentage cover estimates made by BenthoBot were compared to technician estimates for the years 2010, 2011 and 2013 (averaged across all three shoals). Dots plots were used to compare estimates made by the two methods for each benthic category.

In line with output from previous reports, pie charts were constructed to examine broad-scale difference in community composition and represented the proportion of scored points for a given shoal and year. The same nine major categories from the previous report were used to summarise the data (Table 2).

For all other analyses, data were aggregated to major and sub-categories at the Grid Id level (see next paragraph for an explanation) and compared the three shoals, over four sampling years and across two depth bands (shallow: 10 m-30 m; deep >30 m - 50 m).

The long video transects used during the towed video and still image collection were primarily designed to optimise benthic habitat mapping. As the context of the present report aligns more closely with monitoring (observing changes through time) as opposed to mapping objectives, it seemed important to provide analyses based on "replicate" units at a finer spatial scale. One option was to provide analyses at the image level, however image level analysis can often times show high variability, severe zero inflation due to spatial heterogeneity and low point sample size and high spatial autocorrelation. Instead we decided to trial the use of sampling units based on a 50 m x 50 m grid overlayed on each shoal. This provided analysis at a spatial scale more in line with AIMS shallow reef long-term monitoring procedures, which are based on 50 m long fixed transects (Sweatman et al., 2005). Images for each year were grouped by grid, increasing the number of points per sampling unit. The same grid was used at each shoal over time, thus Grid Id could be included as a random effect, creating a pseudo "fixed" transect design (although transects in 2010 were only occasionally overlapped in space with the other sampling years, see Figures 2-4).

Multivariate analyses, implemented using R version 3.0.2 (R Core Team 2014), were used to quantify differences in benthic community composition within and between the shoals. Prior to analysis, percentage cover estimates produced by BenthoBot were aggregated by Grid Id across each shoal, year and depth combination. Data were square-root transformed to stabilize variances, and then used to construct Bray-Curtis similarity matrices. A Principal Coordinates Analysis (PCO) using the function "pco" in the package "ecodist" (Goslee & Urban, 2007) was used to examine differences in community structure between the three shoals, and to explore the influence of year and depth on community type. The "vf" function in the package "ecodist" was used to find the maximum correlation of the individual variables with the first two principal coordinate axes.

Univariate statistical analyses were used to estimate the probability of occurrence of each benthic group for each shoal within each year and depth combination. The number of points scored for each Grid Id for a given benthic category(e.g. soft coral) was modelled as a Binomial distribution, with the number of valid point scores indicating the number of trials (# points per Grid). We used a Bayesian hierarchical model, where Grid Id was included as a random effect, with year, shoal and depth band included as fixed effects, with 95% credible intervals were used to infer differences among factor groups. Univariate models were fit using the INLA package (Rue et al 2009, Lindgren & Rue 2015) in R version 3.0.2 (R core Team 2014).

To explore if Shoal and/or Year (and their interaction) was important, we grouped data by Grid Id and carried out a full subsets analysis of Shoal * Year, with Depth band included in all models. To better understand which of the two factors accounted for more variance, the unique R^2 explained by both year and shoal was calculated from the difference between the model without that variable, and the full interaction model (including depth in all cases).

2.4   Results

The 2016 survey repeated all transects undertaken during the 2011 and 2013 surveys at the three shoals (except Vulcan Shoal where the three transects were omitted due to field constraints in 2013 were also omitted in 2016). While the actual position of the towed video transects may vary laterally by metres or tens of meters, the habitats at these scales should be consistently detected.

Barracouta East and Vulcan Shoal had similar numbers of images analysed over the study (9,170 and 10,114 respectively, Table 3) with the majority of images occurring in the shallow depth band (10 m - 30 m). Goeree Shoal is much smaller and deeper and was not surveyed in 2010. Fewer images were analysed for Goeree (5,976) and the distribution of images was skewed to the deep depth band (>30 m - 50 m).

Table 3. Summary of the number of Images and Grids used to quantify benthic community composition at each combination of Shoal, Year and Depth band. Across the three shoals, 2366 unique Grid IDs contained images. This equated to 4583 Grid IDs across four sampling years (i.e. not all unique Grid Ids contained images in every year). Images <10 m and >50 m were excluded.

Year	Depth	Barracouta East		Goeree		Vulcan	
		Grids	Images	Grids	Images	Grids	Images
2010	<=30 m	173	627			131	462
	>30 m	47	182			59	178
2011	<=30 m	379	2020	65	497	508	2827
	>30 m	132	688	270	1594	146	766
2013	<=30 m	373	2135	62	435	435	2687
	>30 m	122	711	261	1492	79	452
2016	<=30 m	391	2149	66	490	427	2406
	>30 m	121	658	267	1468	69	336
	Totals	**1738**	**9170**	**991**	**5976**	**1854**	**10114**

Total grids 4583

Total images 25260

2.4.1 Comparison of BenthoBot and technician scores

Images scored by BenthoBot showed similar percentage cover estimates to images scored by technicians. In general BenthoBot overestimated biotic categories compared to abiotic categories (Figure 7). Some major categories showed higher consistency between the two methods than others. The confusion matrix (Figure 5) shows how well BenthoBot predicted benthic categories in the training data and is useful in explaining the results in Figure 7. Hard coral was very consistent, and this was supported by the confusion matrix output (Figure 5). Soft coral and other organism estimates were reasonable. Sponge cover estimates were not greatly different but this category had low resolution in the confusion matrix, and low average cover (~ 5%). Although the biotic categories were over-estimated this was a consistent bias across shoals and years, and consequently this bias will not alter general community trends.

Three categories had quite different estimates by the two methods. Algae scored much higher by BenthoBot across all years, consolidated reef much lower and Unconsolidated substrate was both higher and lower. BenthoBot detected seagrass at East Barracouta and Goeree Shoals (Figure 8), whereas technician image analysis had only detected it at Vulcan Shoal. Training images for seagrass consisted of points from Vulcan Shoal represented by flourishing green blades (seen in 2010) and bare rhizomes (seen in 2011). This decreased the capacity of BenthoBot to reliably detect seagrass (bare

rhizomes confused with encrusting red-brown algae or sponge) and was the explanation for why BenthoBot falsely detected seagrass at Barracouta East and Vulcan Shoal. Re-running the confusion matrix on the test-data by Shoal revealed the high f-1 score was due to seagrass predicted well at Vulcan Shoal in 2010 (predominantly green blades) with low f1-scores at Barracouta East and Goeree. So on average seagrass was predicted well, but specifically did poorly at Goeree and Vulcan Shoals. The Grouped category estimates (Table 2) were good for Filter feeders and over-estimated for Primary producers (due to inclusion of algae in this group).
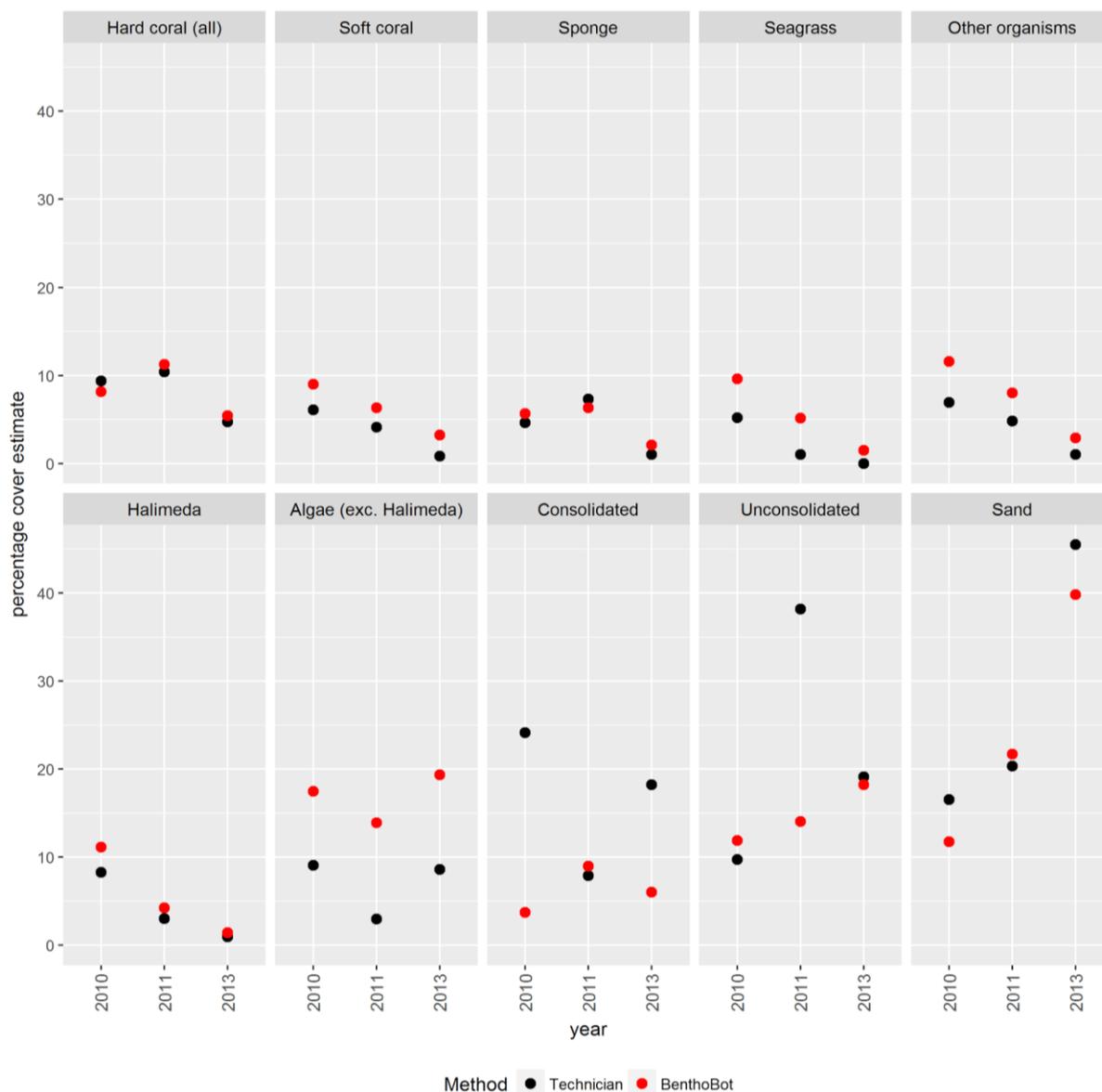
Figure 7. Percentage cover estimates of benthic categories made by BenthoBot were compared to technician estimates for the years 2010, 2011 and 2013 (averaged across all three shoals).

2.4.2 Broad-scale community composition (derived from BenthoBot analysis)

The 2016 survey found that algae in various forms were the most abundant biotic category and this trend was consistent throughout the study period (Figure 8). Hard coral, other organisms and sponges were often the next most abundant groups. Hard coral cover followed a similar trend across all shoals with cover dropping to its lowest in 2013 followed by increases in 2016 (although not as high as

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2010/2011 cover). Goeree typically had the highest hard coral cover, followed by Barracouta East and Vulcan Shoal.

Sponge, soft coral, seagrass and other organisms also reached their lowest cover in 2013, across all shoals, followed by increases in 2016. Decreases in biotic components were predominantly offset by increases in sand (and some unconsolidated substrate such as rubble) at all shoals until 2013. In 2016 the overall trend was for sand to decrease and biotic categories to increase. Sand averaged about 26% cover across all shoals and years but rose to about 40% cover in 2013.
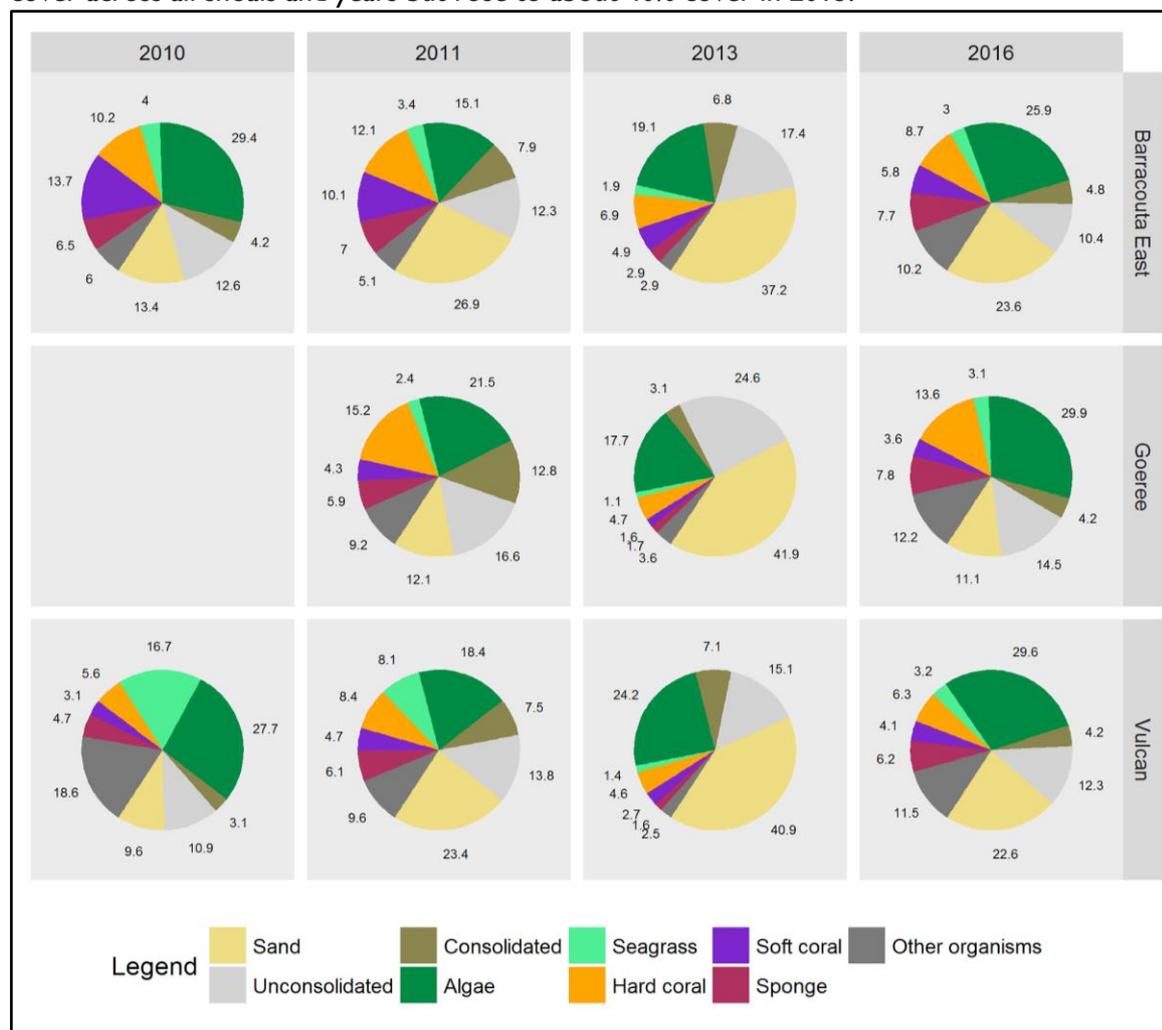
Figure 8. Relative percentage cover of the nine major categories at the three shoals through time based on the BenthoBot analysis.

2.4.3 Temporal changes in benthic categories by shoal, year and depth

Principal Coordinates Analysis (PCO) was used to highlight differences in the benthic community composition between shoals and years. A separate PCO was created for each depth band on the abundances of the 14 benthic sub-categories (Table 2). The shallow PCO (Figure 9) grouped Vulcan Shoal 2010 in a separate cluster characterised by seagrass. The second cluster consisted of all shoals in the year 2013 reflecting low cover for most biotic categories and increases in sand and unconsolidated substrate. The remaining group comprised a combination of 2011 and 2016 shoals with varying contributions from benthic categories. The deep PCO (Figure 10) also grouped shoals together in 2013 associated with low biotic cover (increases in sand and unconsolidated substrate) but was different in that Barracouta East grouped separately from the other shoals in remaining years . The third group consisted of Goeree and Vulcan shoal in 2010, 2011 and 2016.

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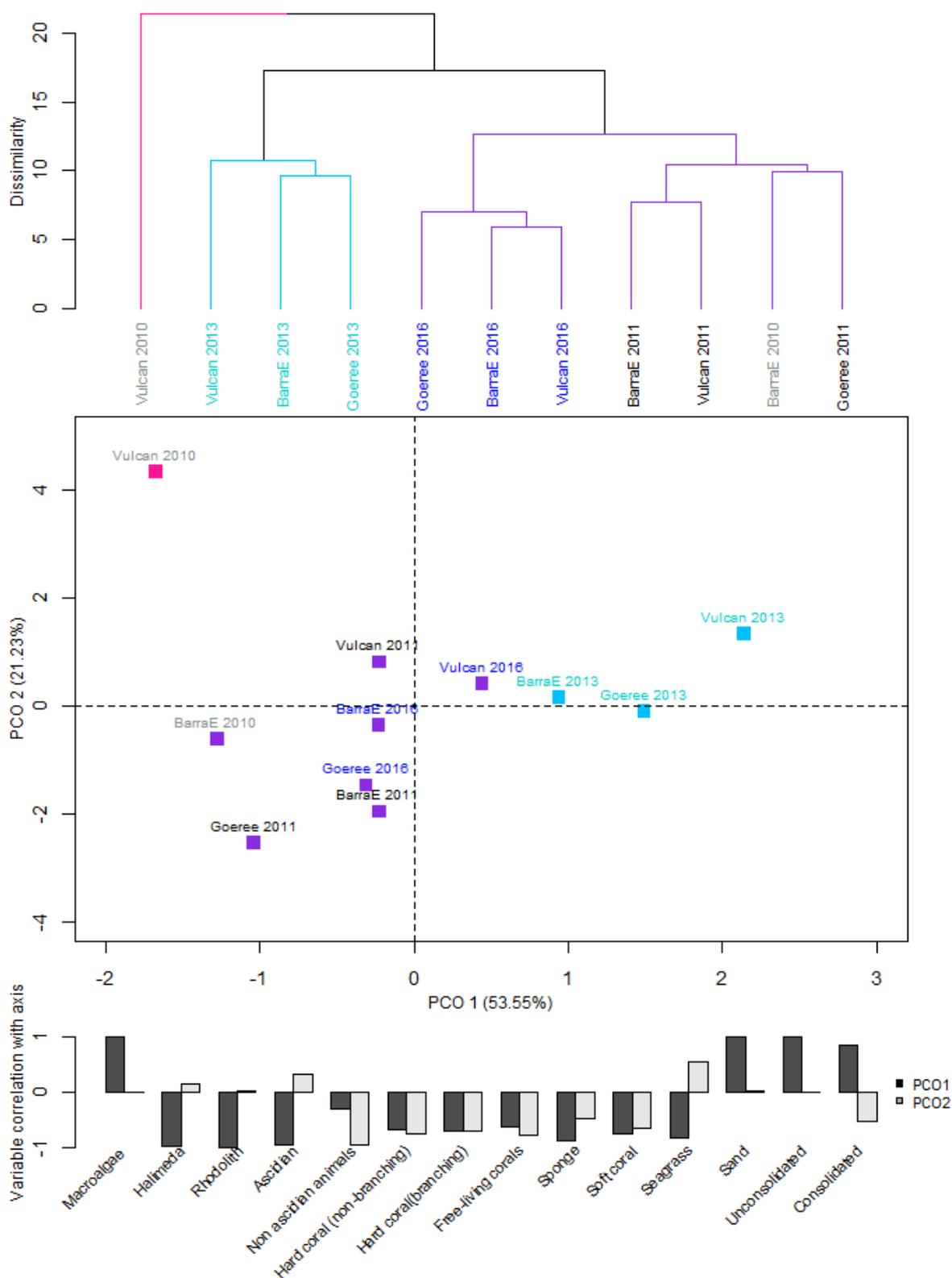
Figure 9. Results from Cluster and Principal Coordinates analysis for the shallow depth band, incorporating the percentage cover of benthic categories by shoal and year. The middle plot is coloured to represent significant groups revealed by cluster analysis (see upper plot) and the lower plot shows the relative contribution of each of the benthic category to the position along the PCO1 and PCO2 axes for each shoal and year combination.

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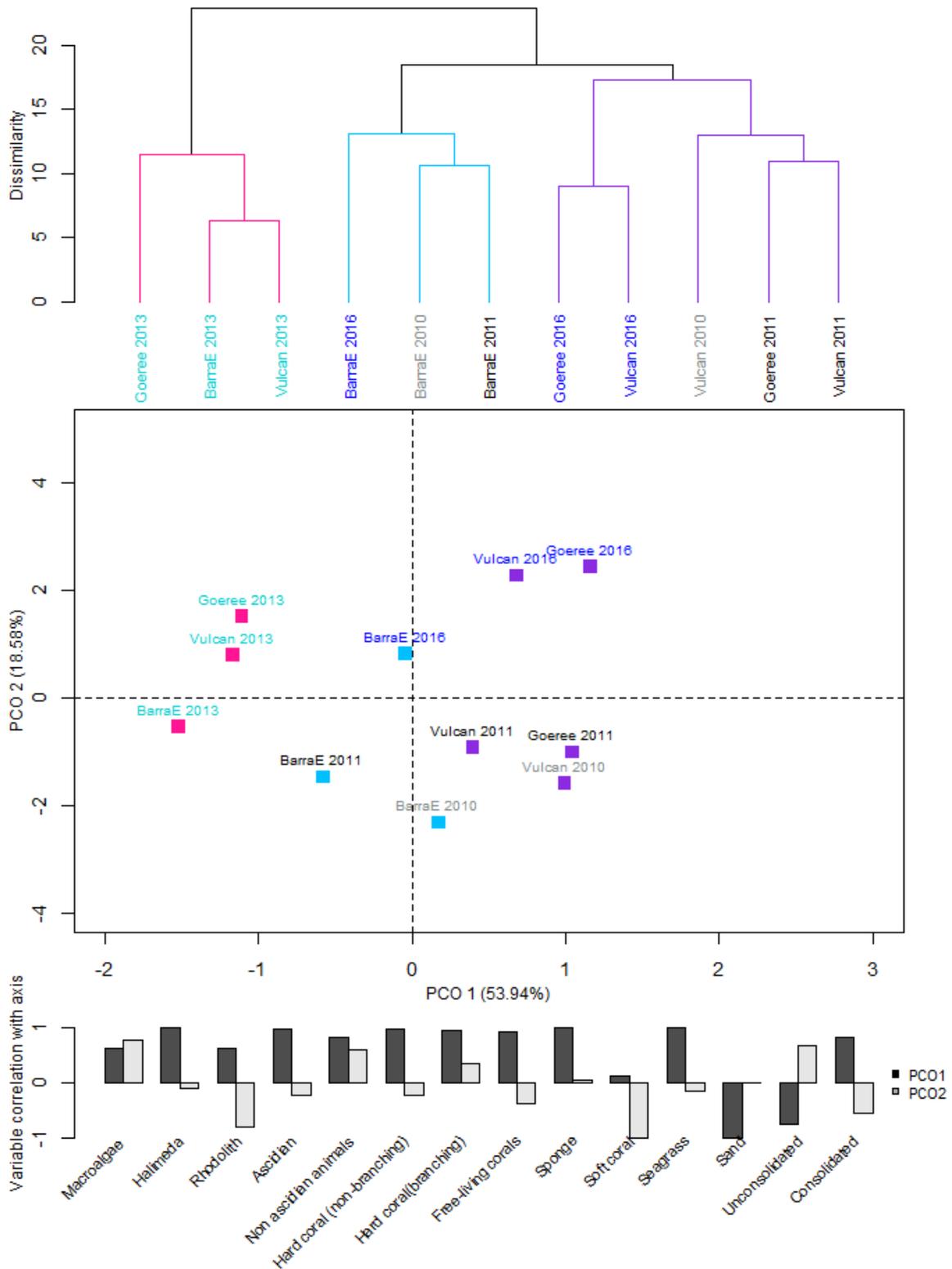
Figure 10. Results from Cluster and Principal Coordinates analysis for the deep depth band, incorporating the percentage cover of benthic categories by shoal and year. The middle plot is coloured to represent significant groups revealed by cluster analysis (see upper plot) and the lower plot shows the relative contribution of each of the benthic category to the position along the PCO1 and PCO2 axes for each shoal and year combination.

The effect of shoal, year, and depth on the relative abundance of the five benthic sub-categories (f1-scores > 0.60, highlighted in red in Table 2) were investigated using Bayesian hierarchical modelling (Figure 11). There was a general trend for groups to decline from 2010/2011 levels to lowest abundance in 2013.

For seagrass (Vulcan 2010, shallow) this drop in cover was greatest in 2011, at which time only leafless rhizomes remained. By 2013 the seagrass meadow was completely gone and there has not been recovery since. Seagrass was only present on Vulcan Shoal (low-levels of seagrass were incorrectly predicted at Barracouta East and Goeree Shoals due to associated image discrimination limitations with BenthoBot). A large noteworthy area of soft coral that dominated the western slope of Barracouta East Shoal in 2010 (deep), dropped slightly in 2011 and markedly by 2013. In 2016, soft coral showed a slight increase in cover and some of the same groups, including *Nephthea* were re-appearing.

A similar but more dynamic trend was seen for *Halimeda* and hard coral which declined in 2011/2013 but showed some recovery by 2016. Most hard coral occurred at Goeree Shoal with branching (shallow) and non-branching groups (both depths) declining in 2013 followed by some increase in 2016. Goeree Shoal was not surveyed in 2010 so it is unknown whether a hard coral decline occurred in 2011 (as seen for seagrass soft coral and *Halimeda* on other shoals).

Grouped categories (see Table 2) comprise all of the five sub-categories and some additional categories that were not predicted well by BenthoBot (such as algae and sponge). The trajectory of hard coral (all) and filter feeders clearly showed the same 2013 drop in cover. Primary producers included all algae and Goeree Shoal (shallow) showed a marked increase in 2013 which was due to increases in macroalgae at this depth (Figure 8), although there was very few images at the shallow depth in any year for Goeree Shoal.
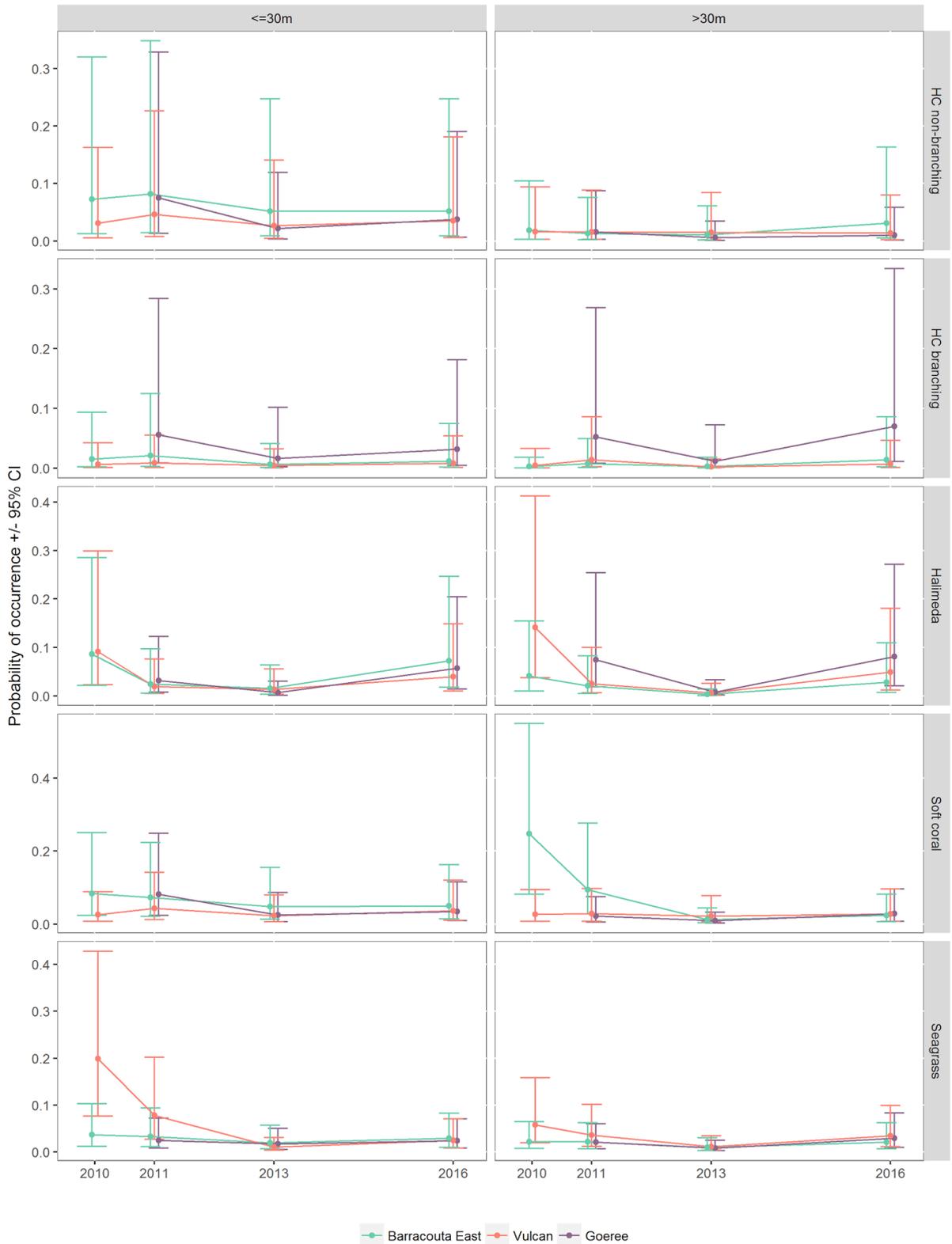
Figure 11. Estimated probability of occurrence (±95% Credible Intervals) of 5 sub-categories at Barracouta East, Goeree and Vulcan Shoal, by year and depth.
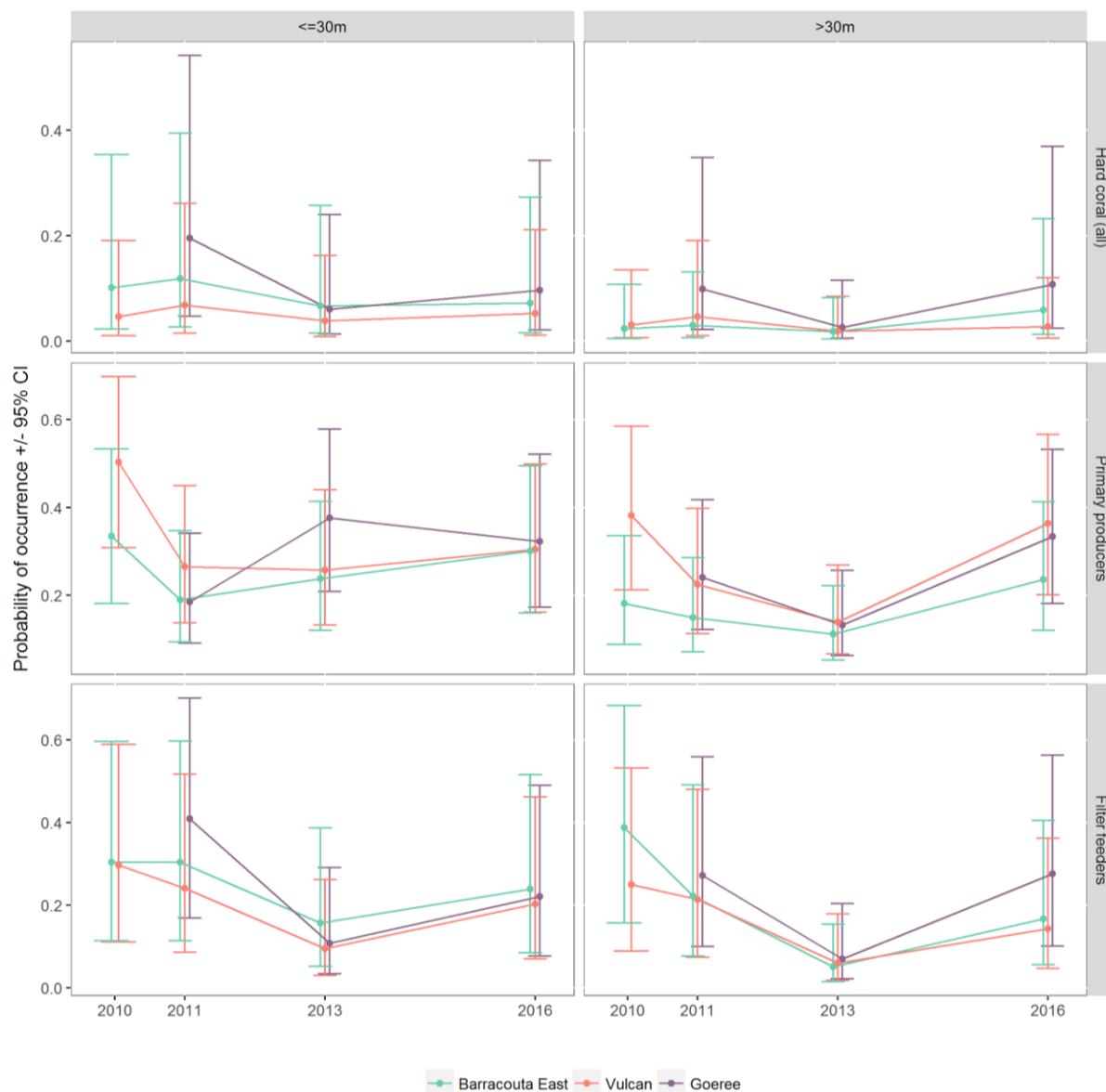
Figure 12. Estimated probability of occurrence (±95% Credible Intervals) of grouped categories at Barracouta East, Goeree and Vulcan Shoal, by year and depth.

We used a full subsets approach to examine the importance of both year and shoal, and their interaction, for the five benthic sub-categories that resolved well by BenthoBot. This analysis showed strong statistical support (AICc model weights equal to 1 in all cases) for a model containing the interaction between year and shoal. To clarify which of the two explained more variance, the R^2 attributable uniquely to either year or shoal was calculated. It showed that for hard coral (branching) and soft coral, the variance explained by shoal was much higher than that explained by year, and for seagrass, year and shoal both have strong effects with year slightly higher. The variance explained by both year and shoal was relatively low for *Halimeda* and hard coral (non-branching) (Figure 13).

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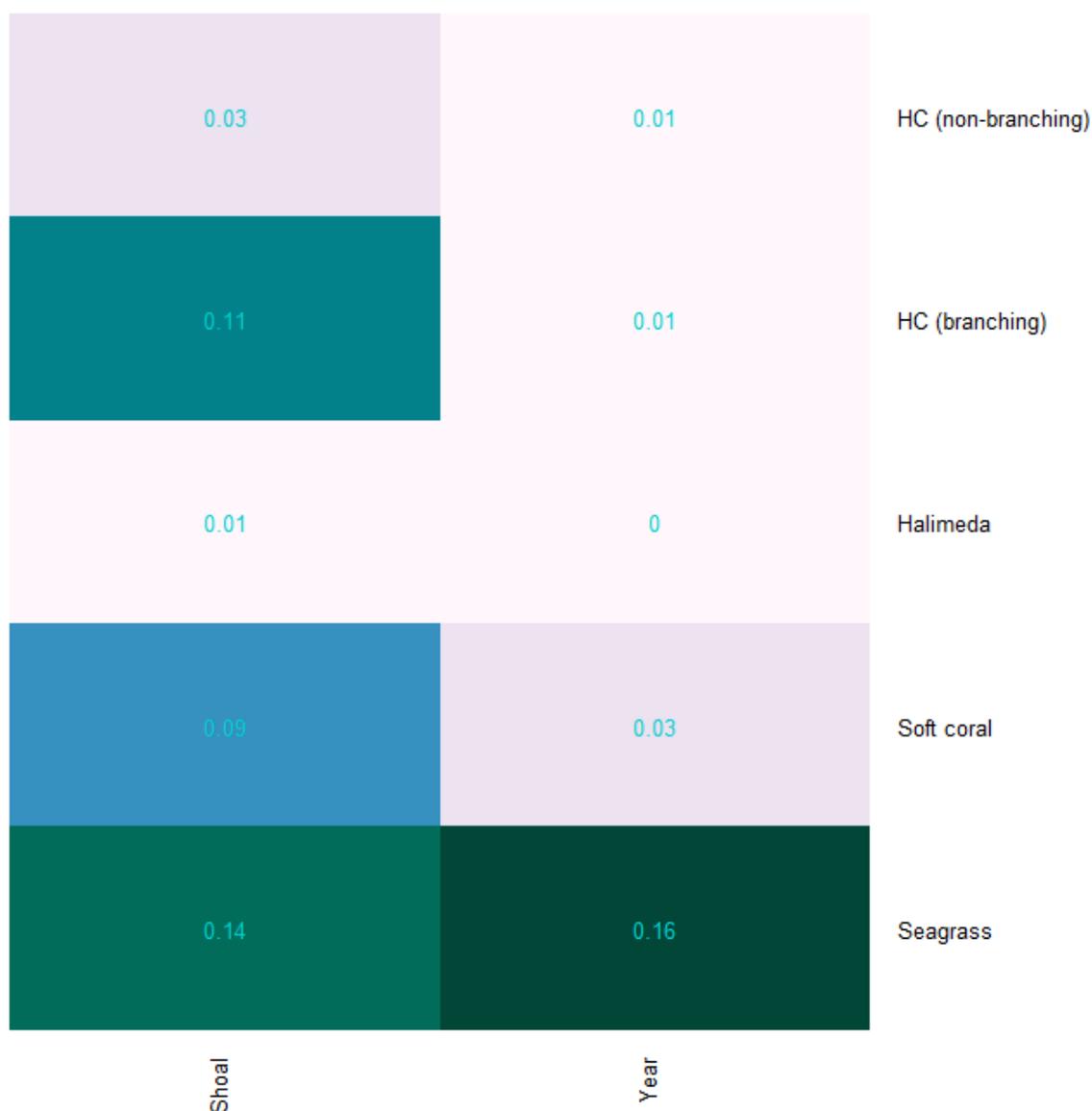
Figure 13. Heatmap of the unique R² value for benthic sub-categories explained by both year and shoal calculated from the difference between the model without that variable, and the full interaction model (including depth in all cases).

2.5 Discussion

2.5.1 Benthic assessment

The 2016 survey provided an additional and important fourth sampling period to the surveys previously undertaken at Barracouta East, Goeree and Vulcan Shoals. In 2013 there was a general trend after the 2011 survey for biota (excluding macroalgae) to decrease across all shoal, year and depth combinations, and for sand and unconsolidated substrate (rubble) to increase. This decrease included hard coral, soft coral, sponge and other organisms (including ascidians). The remaining seagrass at Vulcan Shoal completely disappeared and a soft coral community noteworthy in 2010 and 2011 on the western side of Barracouta East Shoal that had dropped cover slightly in 2011 was lost completely. Puotinen (pers comm) has documented widespread impact to shoals in the region from Cyclone Lua that passed through the area in March 2012. In this study, numerical wave models and field data from

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northwest Australia, show that major coral loss occurred up to 800 km from the track of Cyclone Lua, due to its large size and sustained high wind speeds that generated damaging seas over a vast area.

The 2016 survey indicated a subtle increase in many biota across the three shoals, off-set by a decrease in sand. In general, shoals grouped together in 2010, 2011 and 2016 in PCOs (with the exception of Vulcan 2010 and East Barracouta in the deep depth band). Soft coral increased at Barracouta East Shoal and inspection of images on the western-side of the shoal where the soft coral community had been verified that some of the same genera had re-established in the area. At Vulcan Shoal BenthoBot indicated a slight increase in seagrass cover however manual inspection of the images revealed seagrass had not returned in the area. Dense seagrass habitat has only been detected at two other shoals in the north-west shelf region (pers comm. Heyward) , observed during early Towed video surveys of shoals by AIMS 2003. Those two shoals are adjacent to one another in the Sahul Banks group, located 135 km due north of Vulcan Shoal, at a depth of about 25 m (similar depth to Vulcan Shoal). The seagrass found at those two shoals was the same genera as at Vulcan Shoal, *Thallasodendron spp*. which is an important Indian Ocean species. The distribution and role seagrass plays on shoals of the north-west shelf is not well understood.

Of the three shoals, Barracouta East had medium cover of biotic categories and showed the least amount of change overall through time, although it did show the same trend for biotic categories to reach their lowest cover in 2013. Barracouta East grouped separately in the PCO for all years (excluding 2013) in the deep depth band. Goeree Shoal showed dynamic patterns with fluctuations in hard coral, sponge and *Halimeda* cover. Vulcan Shoal was characterised by the complete loss of its seagrass meadow and has not shown signs of recovery. Our analyses indicated that both shoal and year were important in explaining the variation seen in community composition across all combinations of shoal, year and depth. In summary, benthic communities at the three shoals did not appear to have been directly impacted by the hydrocarbon release in 2009, whereas physical disturbance associated with storms may be an important driver of shoal communities.

2.5.2 Classification of images by BenthoBot

The process of training BenthoBot, building the model and validating classifications has highlighted some broader issues with classification of benthic categories. Categories created by technicians were used to train BenthoBot and build the model. These categories focus on biological/ecological aspects of the benthic community without regard for how well these might be resolved by the measurable properties of two dimensional digital images. An alternative approach worth investigating in future studies would be to reverse this process, i.e. allow BenthoBot to generate its own groups based on spectrums, textures and shapes that are able to be clearly resolved, and to then assess the usefulness of these categories in the context of the biological and ecological relevance. Where there are benthic groups of high interest, i.e. indicator groups (typically hard coral, some sponge and algal groups, soft coral, crustose coralline algae), these should continue to be identified to the highest level and will require additional classification by technicians.

Providing training images for a survey region with accurate and consistent classifications is fundamental to fine-tuning BenthoBot. Factors that reduce the accuracy with which points have been classified - such as classification by numerous technicians, bad image quality, or benthos that is hard to distinguish (i.e. similar colour spectrum and features) - also reduce the accuracy of BenthoBot. Some categories, such as seagrass consisted of two vastly different states that may have resolved better if their original scoring by technicians had been split into more than one group, i.e. seagrass (green blades) and seagrass (rhizomes).

Inter-observer differences in benthic habitat classification, transfers as reduced precision of the BenthoBot model to predict categories, i.e. a model is as good as the data it is trained on. In a recent

publication by Beijbom *et. al.* (2015) an analysis was done comparing performance of computer classification (i.e. similar to BenthoBot) against multiple technicians of the same datasets. It found strong agreement between technicians for hard coral (79.68-86.5%) but very low agreement for macroalgae (48.29-65.54%), coralline algae (22.64-52.68%) and turf (28.89-53.92%). Overall the computer classification for high level categories was more consistent at categorising benthos than a technician. These results align with the results seen in this study in relation to well resolved benthic categories and those which were poorly predicted, with the poorly predicted categories typically inconsistently identified by the technicians. This may be improved with more refined datasets for training purposes, that only use data with multiple observer agreement. Other techniques to improve accuracy may involve adding extra parameters to the model during training, for example depth, slope and rugosity measures, and using multiple patch sizes.

2.6   Recommendations

This study confirms that the shoals support diverse benthic communities, with the plateau regions receiving enough light at the seabed to support benthic primary producers and many species typical of tropical coral reef ecosystems. Major habitat types on the shoal plateaus have persisted over the period 2010-2016, but the abundance of key biota has varied in time. Marked declines have been observed in selected organisms such as seagrass on Vulcan Shoal and soft coral at Barracouta East Shoal. Such changes have occurred at different times and the causes of these disturbances to the benthos remain unclear, although hydrodynamic forces associated with wave energy could be contributing to mobilisation of unconsolidated substrates. Other organisms such as hard corals have shown more moderate changes, declining then increasing over the study period. Both declines and increases in major benthic categories, such as hard corals, other primary producers and filter feeders were recorded between successive surveys at all three shoals, consistent with the observed temporal variability being unrelated to distance from the MWHP.

The shoals support a range of biota that may respond differently to both natural and human pressures. Broad-scale monitoring using towed camera has proved effective at delineating the boundaries of major habitat transitions and will remain a useful tool for rapid overview of the status of key organisms and habitat types on each shoal. In combination with rapid image classification, via tools such as BenthoBot, general monitoring in the future will become more cost effective.

Adaptive monitoring of shoal habitats is likely on the near horizon. Even at its present level of development Benthobot, if provided with updated field imagery, would be capable of detecting a major departure in the abundance of shoal biota from the status quo, which may then trigger a more resource intensive assessment. Such an approach to monitoring could be combined with more spatially precise repeated measures of selected habitats, using existing acoustic-tracked ROV equipment, to maximise the ability to detect temporal change. Repeated fine scale multibeam survey shoal geomorphic surveys can also be used as a method to detect and determine hydrodynamic disturbances on these shoals caused for example by storms. These would be useful in verifying both the lightly cause and spatial extent of substrate loss which is the foundation for establishment and maintenance of all major benthic communities (including seagrass, coral, macroalge and filter feeders). It would be desirable, should future monitoring on these shoals be required, to establish monitoring locations in a range of key habitats that may respond to natural or human pressures differentially. These selected habitat areas, which can be identified suing towed camera systems, would include areas dominated by hard corals, soft corals and selected plants such as *Halimeda* and seagrass if present. Repeated spatially precise sampling within habitat types would provide an improved sensitivity to detect changes in selected organisms, rather than rely on averages estimated from all data across each shoal plateau. For example across an entire shoal, hard coral cover may average only 8% (as it includes large areas of sand) making its average contribution to benthic cover low, whereas within hard coral habitats hard coral abundance can reach over 40% cover. Repeated monitoring within the coral habitat would allow precise

quantification and maximum discrimination of changes specific to corals, which may be at variance with changes in other sensitive receptors. The BenthoBot classification used for the first time in the study, supports this approach by providing a broad-scale and shoal-wide assessment of the benthos, whilst identifying key habitats that require further fine-scale classification by technicians.

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3. Baited Remotely Video Systems (BRUVS™)

3.1   Introduction

Submerged shoals and banks at various distances from the MWHP were surveyed in 2011 to assess the potential impacts of uncontrolled release of hydrocarbons in 2009 on the associated fish communities. Sites were originally sampled as part of the Montara Shoals project, and while a baseline was not established, data from three surveys (each separated by two years) across three shoals was collected for comparison. This report investigated fish species richness and diversity with associated habitat through time, at three shoals reported in surveys conducted in 2011 (Heyward *et al.* 2012), 2013 (Heyward et al. 2013), and 2016 (this report). The aim was to determine whether fish communities displayed any inter-annual (2011, 2013 and 2016) or spatial (Barracouta East, Goeree and Vulcan shoals) patterns, specifically to identify effects of year of sampling on species richness, relative abundance, and fish length compositions.

3.2   Methods

3.2.1   Field sampling with stereo Baited Remote Underwater Video Systems (BRUVS™)

Demersal fish communities were surveyed using stereo BRUVS™ (Cappo et al. 2007, Harvey et al. 2002, Watson 2006) following 2011 and 2013 surveys (Heyward et al. 2012, 2013). Heyward et al. (2012) provides a comprehensive summary of the design, use and calibration of BRUVS™ (and references therein), but briefly, stereo BRUVS™ consist of two camera housings on a light galvanised steel frame with a bait pole projecting into the field of view. The bait pole attached from front side of the frame carried a flashing diode (to synchronise video frames) and a bait bag, which contained 1 kg of crushed pilchards (*Sardinops sagax*) at a distance of approximately 1.2 m in front of the video housings (Figure 3.1). The frame of the BRUV was ballasted with weights for stability in currents and waves, and the system was retrieved after deployment via a rope attached to surface buoys (Figure 3.2).
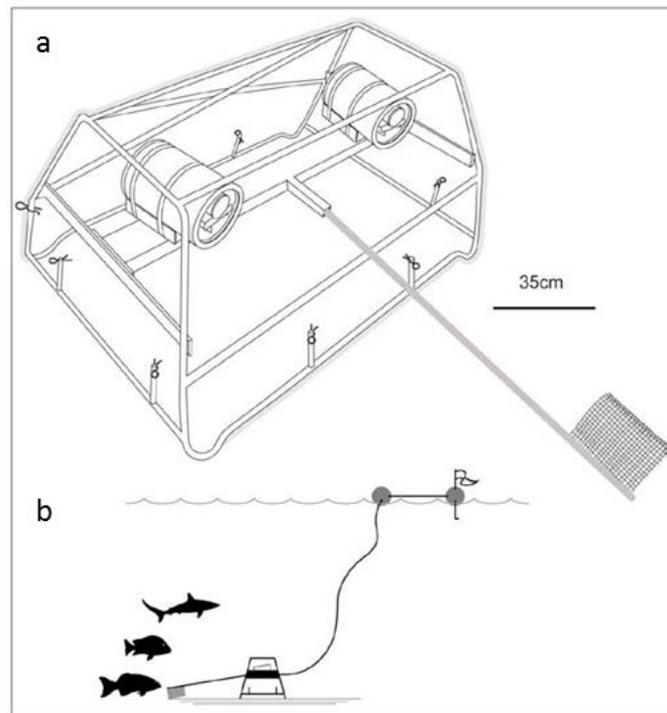
Figure 3.1. A stereo BRUVS™ unit with bait arm attached (a) showing typical deployment with bait bag touching the seabed (b).
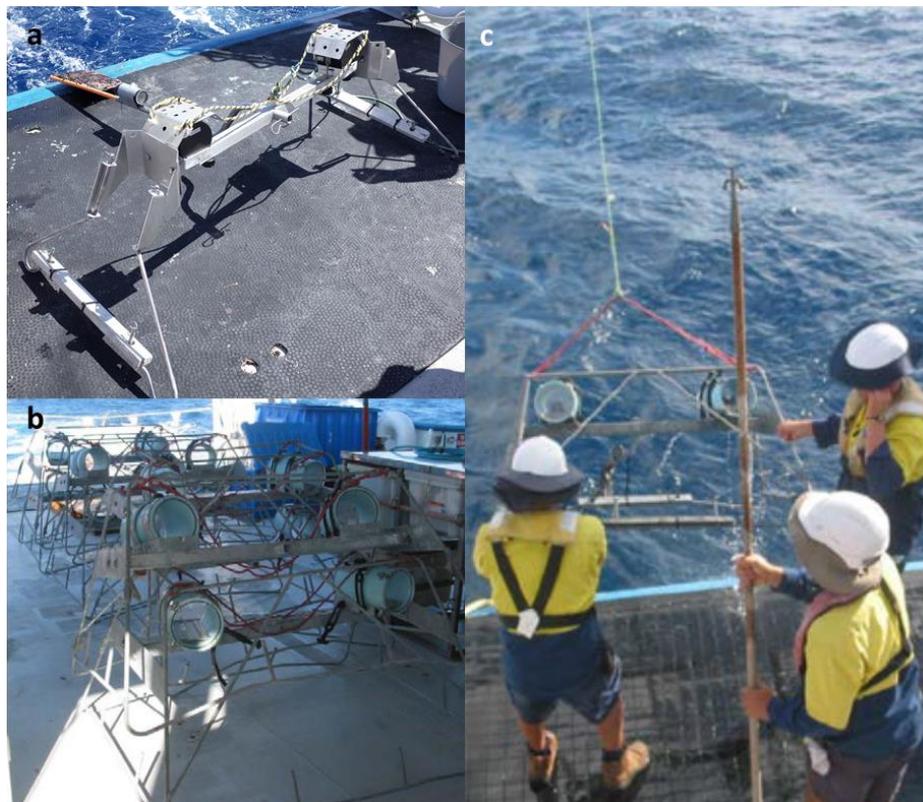
Figure 3.2. Stereo BRUVS™ units ready for deployment, and during the process of retrieval.

During the 2011 and 2013 surveys, Sony HDR-CX110E 'handycam' cameras (x 0.6 wide conversion lens) were used within each housing. Cameras were set to record at 1920 x 1080 pixels (high definition), with focus set to infinity in manual focus mode. Each camera recorded to a 16GB SD

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memory card that was downloaded upon retrieval of the BRUVS™ and stored on portable hard drives in .m2ts file format.

GoPro Hero4 Silver cameras were used for the 2016 survey with recording set at 30 frames per second, 1920 x 1080 pixel resolution, and medium field of view. Each camera recorded to a 32GB microSD memory card that was downloaded and stored on portable hard drives in .MP4 file format.

Standardised, repeated samples were taken with SBRUVS at the three shoals at varying distances from the uncontrolled release for each survey; Barracuda East (B) categorized as low exposure, Goeree (G) categorized as moderate-high exposure and Vulcan (V), categorized as high exposure (Table 3.1; Heyward *et al.* 2012). Barracouta East Shoal (n = 24 sites), Goeree Shoal (n = 24 sites) and Vulcan Shoal (n = 24 sites) were surveyed during March and April (autumn) in 2011 and 2013, and in September (spring) of 2016.

Sampling for 2013 and 2016 followed the location of sampling sites for the deployment of BRUVS™ as closely as possible (Figure 3.3). Comparisons in habitat composition and depth among 2011 and 2013 deployments revealed no significant variation among deployment sites (Heyward et al. 2013). BRUVS™ were deployed above the 60 m depth contour, with a minimum distance of 250 m between deployments. During a single day, a total of 24 deployments of BRUVS™ occurred on each shoal. A 60 minutes minimum soak time was used for each BRUVS™ deployment, with time based on species accumulation curves from surveys of reef fish faunas in the north-western atolls (Scott Reef) by Cappo *et al.* (2001) and the distance between deployments (250 m) avoided potential overlap of bait plumes and the movement of fish between BRUVS™ (see Cappo *et al.* 2004 for review).
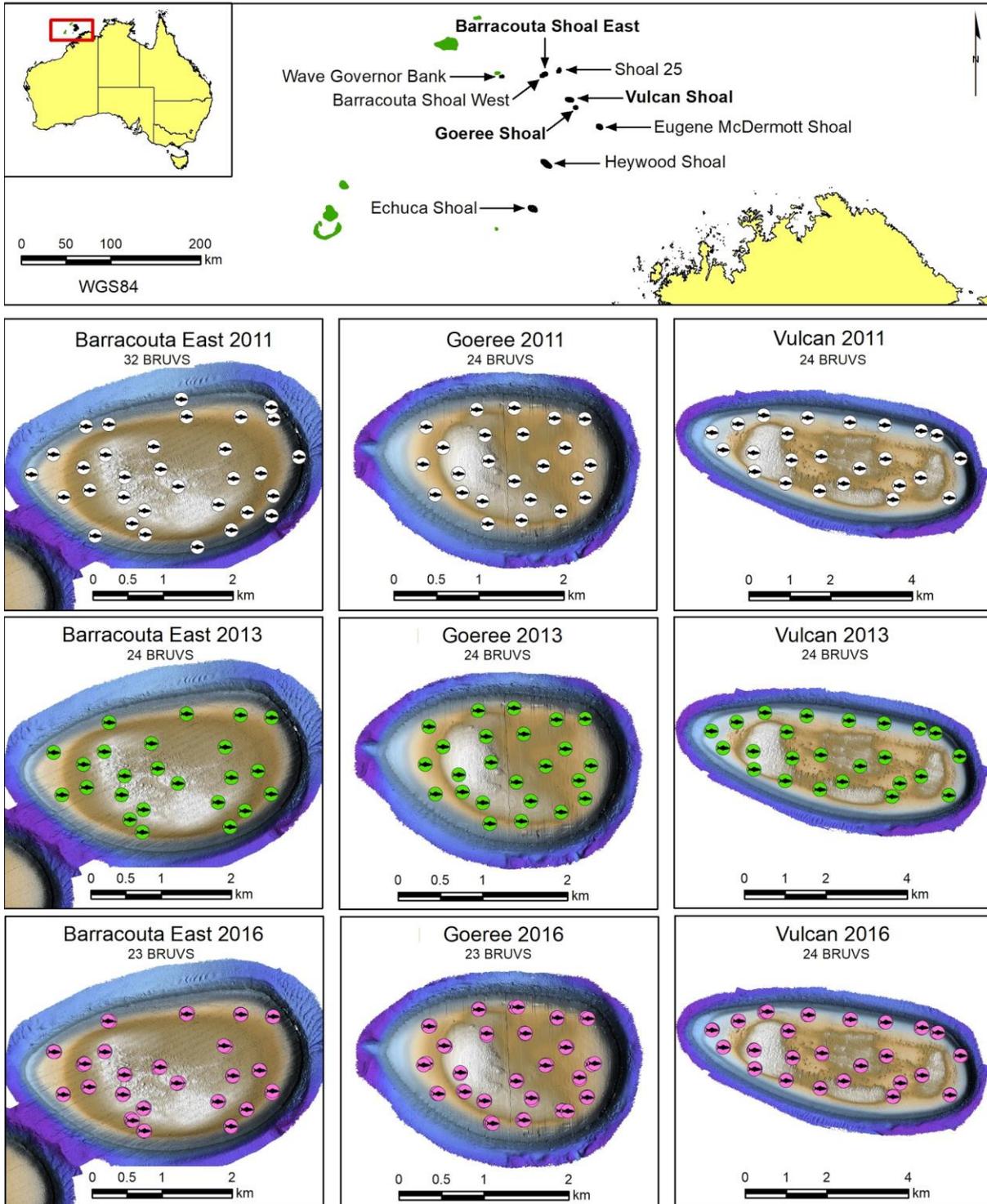
Figure 3.3. Deployment sites for stereo BRUVS™ at three shoals in 2011 surveys (upper panels), 2013 (middle panels), and 2016 (lower panels).

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Table 3.1. Three shoals sampled in each survey with abbreviations (), the number of BRUVS™ samples per shoal and bank (n) in 2011 and 2013, location (mean latitude and longitude), the linear distance (spilldist) and compass bearing from the uncontrolled release, the sediment hydrocarbon concentration [HC] with number of sediment samples in (), the minimum and maximum hours of modelled exposure (minHRS, maxHRS), the area of the shoal above 60 m depth and the mean depths of BRUVS™ samples in each survey.

	Barracuda East (B)	Goeree (G)	Vulcan (V)
n 2011	23	23	24
n 2013	24	24	24
n 2016	24	24	24
Mean latitude	-12.55	-12.882	-12.802
Mean longitude	124.03	124.3427	124.2816
Exposure cat	Low	High	High
Spilldist (km)	56.56	31.65	31.42
Bearing(o)	284.47	222.33	242.42
[HC] µg/g	0.015 (4)	0.144 (4)	0.241 (4)
minHRS	13	424	473
maxHRS	25	849	945
Area(ha)	603	320	1299
Mean depth 2011 (m)	27	35.5	29.7
Mean depth 2013 (m)	26.7	35.3	29.5
Mean depth 2016 (m)	26.2	35.9	28.8

3.2.2 Fish community and habitat data

Fish community data

At each deployment site, fish communities were characterised with respect to the diversity, total abundance and size of key species observed. Video analysis of all samples collected by stereo BRUVS™ followed the same standards described in detail in Heyward et al. (2012, 2013). Briefly, for each 60 minute deployment, the maximum number of fish belonging to each species that were present in the field of view at one time period (MaxN) was used as the measure of relative abundance for analysis (Priede et al. 1994, Cappo et al. 2004). Records were made for each species, of the maximum numbers seen together in progression of the whole tape (MaxN) and updated times at which each MaxN occurred. The use of MaxN as a common metric of relative abundance has been reviewed by Schobernd et al. (2014) and Willis and Babcock (2000).

The term 'fish' refers to any marine vertebrate seen in the field of view, including sharks, rays and sea snakes. Individual fish were identified to the highest taxonomic level possible using available literature (Randall et al. 1997, Allen et al. 1998, Lieske and Myers 2001, Randall 2002, Allen et al. 2003, Allen 2004) and the UWA and AIMS image and video reference libraries. Common and scientific names follows those reported in Allen & Swainston (1988) and the codes and conventions of the Commonwealth codes for Australian aquatic biota (Rees et al. 2011) were followed for the naming of species. All unidentified taxa (identified only to genus) were included in analysis, in which fishes from the same genus that could not be identified to species were labelled as *Genus* sp1 to *n*, where *n* was the total number of unknown but distinct taxa in the genus. Likewise, if a species could be recognised as unique, but not identified to either species or genus, it was labelled as *Family* sp 1 to *n*. Fishes that were difficult to identify on video footage at depth, and were subject to variability in identification by readers over the three years of video analysis were grouped for statistical analysis (Table 3.2). Recent taxonomic changes (Last et al. 2016) during the course of the surveys resulted in renaming *Dasyatis kuhlii* and *D. australiae* to *Neotrygon australiae*.

Table 3.2. Species combined into species groups for richness and abundance analyses.

Species Group	Family	Species
Plectropomus group	Serranidae	*P. leopardus P. laevis, Plectropomus sp.*
Cirrhilabrus group	Labridae	*Cirrhilabrus exquisitus, C. punctatus, C. temminckii, C. randalli, C. sp, C. sp1, C. sp2, C. sp, C. spp*
Scomberomorus group	Scombridae	*Scomboromorus commerson , S. queenslandicus, S. sp, S. spp*
Variola group	Serranidae	*Variola albimarginata, V. louti, V. sp*
Lethrinus olivaceus/microdon	Lethrinidae	*Lethrinus olivaceus, L. microdon*
Acanthurus group1	Acanthuridae	*Acanthurus bariene, A. dussumieri, A. mata, A. xanthopterus*
Acanthurus group2	Acanthuridae	*Acanthurus grammoptilus, A. blochii, A. nigricauda*
Scarus oviceps/forsteni group	Scaridae	*Scarus forsteni, S. oviceps*
Haliochores zeylonicus group	Labridae	*Leptojulis cyanopleura, Coris pictoides, Halichoeres zeylonicus, H. hartzfeldii*
Chromis fumea/amboinensis group	Pomacentridae	*Chromis fumea, C. amboinensis*

The fish community structure dataset contained abundance of each species and species richness for deployments repeated at each site over the three survey years. Caudal fork length was also measured where possible and field of view was unobscured for 6 key species for specific analysis. These species included the spot cheeked emperor *Lethrinus rubrioperculatus*, yellowtail emperor *L. atkinsoni*, ambon emperor *L. amboinensis*, long nose emperor *L. olivaceus*, red bass *Lutjanus bohar*, and white-margined coronation trout *Variola albimarginata*. Selection of the six key species was based on their abundance, easy identification, presence on all shoals, and carnivorous diet. These species are of economic importance in the Indo-Pacific region as targets of commercial and/or recreational fisheries and are classified as important meso-predators.

Habitat data

Habitat data consisted of information on the seabed composition (proportion abiotic cover), topography and nature of epibenthic plant and animal communities (proportion biotic cover). These data were generated from the BRUVS™ field of view. Some data were also derived from previous multibeam surveys where these had been incorporated in previous fish/habitat models (i.e. profile to predict mean fish length; Heyward et al. 2012). This study used the standardised classification scheme for the seabed in the BRUVS™ field of view produced by Heyward et al. 2012 (see Table 4.2 in Heyward et al. 2012).

3.3   Data Analysis

The 2016 survey was the third assessment undertaken for the PTTEPA project and this report included data from all three years (2011, 2013, 2016) for comparison. Descriptive statistics derived for the each survey included determination of numbers of species and families and the calculations of means and standard errors for key species size.

3.3.1   Temporal changes in species richness and abundance with shoal and habitat

Temporal and spatial trends in fish communities were analysed in three ways. For each analysis, richness and abundance were modelled as response variables, with year and shoal modelled as fixed factors. Numbered sites where individual BRUVS were deployed in repeated years were treated as a random effect to account for the lack of temporal and spatial independence among individual deployments (repeated measures). An observation level (individual BRUVS deployments) random term was also included to account for over dispersion.

First, richness and abundance were modelled as a Poisson distribution using the Bayesian hierarchical model as per the benthic analyses. At each shoal within each year and depth combination (shallow: <= 30 m; deep: > 30 m). Differences among factor groups band were inferred using 95% credible intervals. Univariate models were fit using the INLA package (Rue et al 2009, Lindgren & Rue 2015) in R version 3.1.0 (R core Team 2014).

Second, the influence of shoal, year and the interaction between shoal and year on richness and abundance of fishes was tested using Generalized Additive Mixed Models (GAMM) fitted by maximum likelihood (Laplace Approximation). To understand the variance explained by year and shoal, the unique R^2 explained by both year and shoal was calculated from the difference between the model without that variable, and the full interaction model (including depth modelled as a smoothed predictor). Models were analysed using the *gamm4* package in R (Wood and Scheipl 2016) using a Poisson distribution and a full-subsets information theoretic model selection process based on AIC_c (Burnham and Anderson 2003).

Using an information theoretic procedure, we fitted all possible combinations of models up to three included predictors, excluding any models were predictors were correlated by more than 0.28. First order interactions between continuous predictors and the year and shoal factors were also included. To determine the best fitting model for both richness and relative abundance, models were compared using the Akaike Information Criterion corrected for small sample bias (AICc). Strongly supported models were those with ΔAICc values < 2 of the minimum, and relative model weights were also calculated (Burnham and Anderson 2003). Where multiple models satisfied this criterion, the best-approximating model was considered as the model with the lowest ΔAICc and the fewest terms, and the proportion of variance explained by the fixed and random factors was assessed.

Third, GAMMs were fit by maximum likelihood (Laplace Approximation) without the effect of depth, to determine whether fish richness and abundance were influenced specifically by time, shoal and habitat predictors. Depth was treated as a null term (smoothed predictor) in all models because it is well established that fish communities are influenced by depth (Chatfield et al. 2010, Harvey et al. 2013, Asher et al. 2017). Spatial parameters as continuous smoothed predictors and included biotic cover (proportion of hard coral, plants, bare substrate), abiotic cover (proportion of boulder, calcareous reef, gravel, rubble, sand), and aspect (a multibeam derived descriptor, identified as important to species metrics previously; Heyward et al. 2013).

3.3.2 Temporal changes in fish communities

Each shoal was analysed separately with species present on at least 5% of samples (four occurrences for the 66 samples from each of Barracouta East and Goeree shoals and 71 Vulcan shoal samples with useful data). The species abundance (raw scale) by shoal matrix was standardised using the Hellinger method, which is defined as:

$$y'_{ij} = (y_{ij}/y_{i.})^{0.5}$$

Where *j* indexes the species, *i* the site/sample, and *i.* is the row sum for the i_{th} sample.
In contrast to transformation, standardisation implies that the entries are transformed relative to other entries. Redundancy analysis (RDA) was conducted for each shoal using the standardised data matrix using the fixed effect of year, conditioned by site. The Condition() term defines partial terms that are fitted before other constraints and can be used to remove the effects of background variables.

Site scores, weighted averages, and centroids from ordinations of the multi-dimensional response into two dimensions were produced in the *vegan* package (Oksanen et al. 2017; *rda* function). The model enabled partitioning of the multivariate species variation explained by year, and by site, and the unconstrained, unexplained variation.

Direction of species abundance vectors (in the k-dimensional ordination space) that had maximal correlation with predicting year of sampling was achieved via *envfit* in *vegan*. A threshold correlation of $p < 0.001$ was set to select significant species vectors for biplots.

3.3.3 Temporal and spatial differences in key species length frequency distributions

For each of the 6 key species (*Lethrinus rubrioperculatus*, *L. atkinsoni*, *L. amboinensis*, *L. olivaceus*, *Lutjanus bohar*, and *Variola albimarginata*), estimated cumulative density functions (ECDF) were plotted over length frequency histograms, and the effect of shoal and year with fork length was tested using anovas with multiple comparison tests (using *multcomp* package; Hothorn et al. 2016) to identify significant ($p < 0.05$) pairwise comparisons.

3.4 Results

A total of 214 BRUVS™ deployments from the three shoals were analysed with 20,627 fishes, representing 50 families and 362 species (Table 3.3). Teleosts comprised 96% of individuals (19,798) observed, with 3.5% (721) elasmobranchs and 0.5% (108) sea snakes recorded across the shoals. At each shoal, the total number of species observed was greater in 2013 compared to other years, while total abundance varied among shoal and year (Table 3.3).

Table 3.3. Fish diversity indices for BRUVS deployments by shoal (Barracouta East, Goeree, and Vulcan shoals) by year (2011, 2013, 2016). The "Unique species" row refers to the number of species seen only in one cell of the shoal by year matrix. SBRUVS summarises the number of video files analysed from each sampling event.

	Barracouta East			Goeree			Vulcan			
	2011	2013	2016	2011	2013	2016	2011	2013	2016	Totals
SBRUVS	23	24	24	23	24	24	24	24	24	214
Families	32	36	34	32	33	33	35	37	31	50
Genera	78	80	82	79	89	85	84	91	77	145
Species	149	159	151	145	165	161	142	165	137	362
Individuals	2919	2666	2662	1970	2348	2527	2032	1670	1833	20627
Unique species	9	17	15	14	18	19	7	14	10	

Overall, richness and abundance of fishes were variable among shoal and year, with greater variability in fish communities observed among shoals than with years (Figure 3.4). While over 60 species were observed at some sites at Goeree shoal, species richness was consistently higher at Barracouta East shoal across all years. The effect of individual BRUVS deployment sites were not considered in Figure 3.4, and overlapping 95% confidence intervals between years within shoals does not necessarily indicate lack of statistical significance. Across the three years, fish abundance was relatively consistent with no clear trend observed among shoals and years (Figure 3.4).
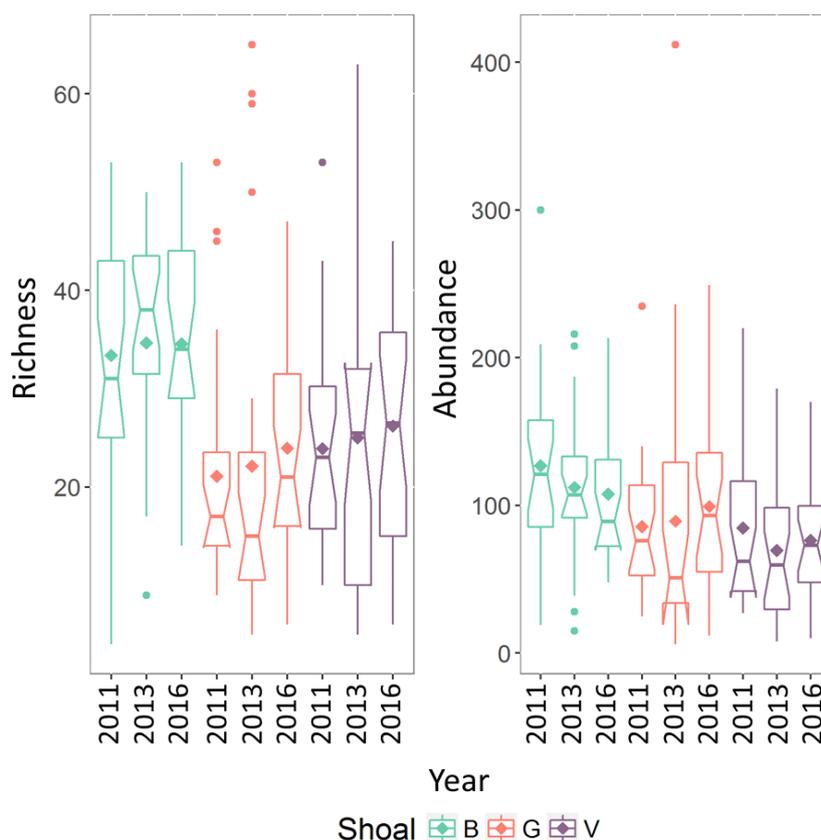
Figure 3.4. Boxplots of interquartile ranges and outliers in fish species richness (a) and abundance (b) by shoal (B: Barracouta East; G: Goeree; and V: Vulcan shoals) over 3 years (2011, 2013 and 2016). Notches indicate ~95% Confidence Intervals for comparing medians, means symbols within the boxes.

3.4.1 Temporal changes in richness and abundance with shoal and habitat

The effect of shoal, year, and depth (shallow: <= 30 m; deep: > 30 m) on species richness and abundance were investigated using Bayesian hierarchical modeling. Two outliers with total abundances of 412 and 300 individuals were excluded from the dataset to avoid skewing model results.

Fishes were more speciose and abundant at shallow sites (<= 30 m) with greater variability than observed in deeper sites (> 30m; Figure 3.5). The model reported greater richness at Barracouta East shoal, consistent with observed data (Figure 3.4), particularly at shallow depths. Likewise, high abundance values were predicted for Goeree shoal in 2013, which appears attributed to fishes occurring in shallower depths. Aside from these trends, richness and abundance was relatively consistent across years (Figure 3.5), with a greater difference in metrics among shoals observed overall.
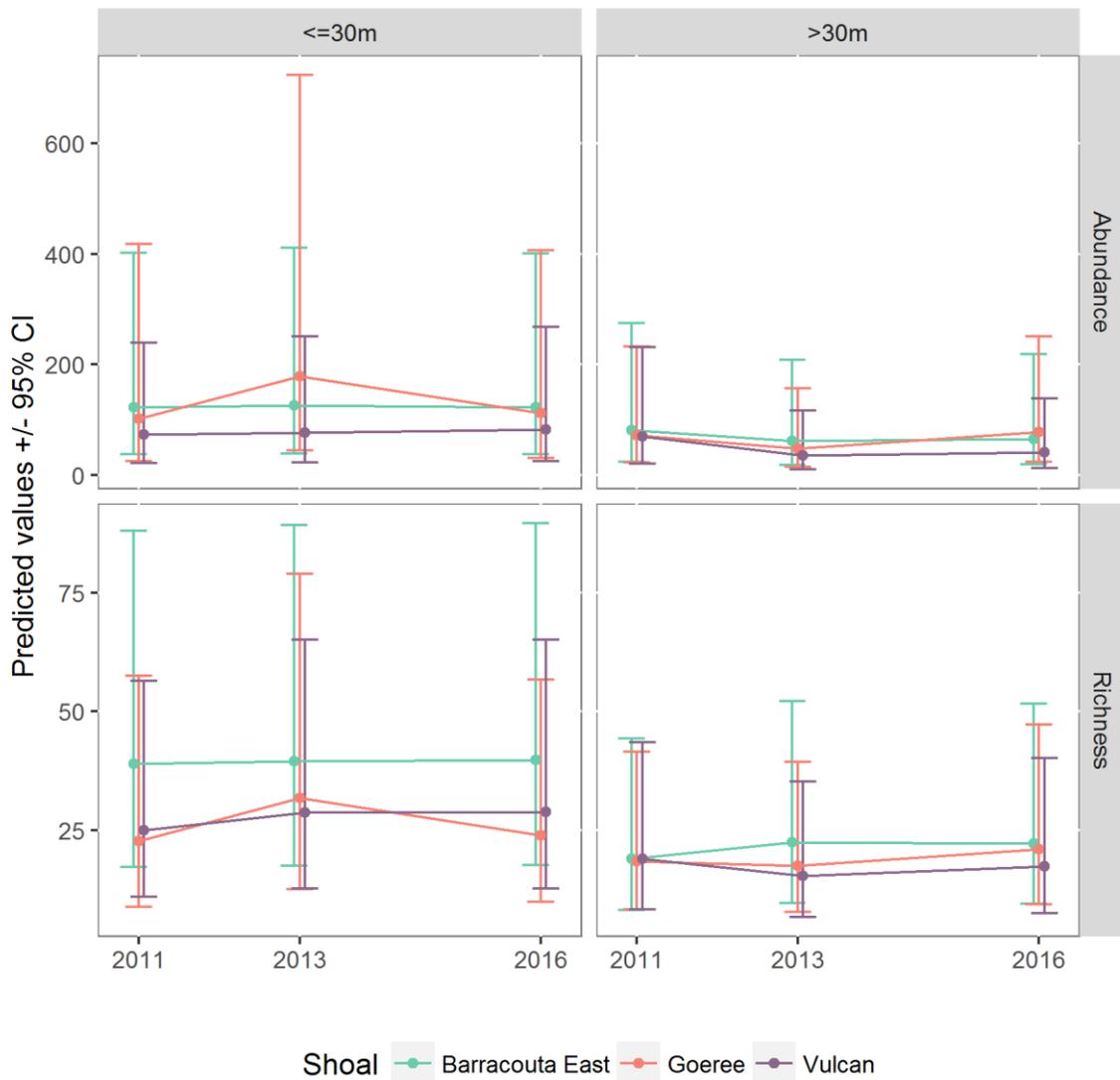
Figure 3.5. Bayesian model predictions of richness and abundance by year and shoal with depth category.

GAMMs used to test for year and shoal effects that included depth (smoothed predictor) produced higher r^2 values for shoal in comparison to year (Figure 3.6). It appears there is little effect on the fish community observed through time attributed to year.

Figure 3.6. Estimated Model predictions of richness and abundance (±95% Credible Intervals) at Barracouta East, Goeree and Vulcan Shoals, by year and depth.

The effect year and shoal, biotic habitat (hard coral, plants, bare cover), biotic habitat (boulder, calcareous reef, gravel, rubble), and a spatial habitat derivative (aspect) on richness and abundance were assessed using GAMM models, without the effect of depth (null term). Three models (R1-R3) were strongly supported by the richness data (ΔAICc < 2 criteria), with models comprising various combinations of four predictors: calcareous reef habitat (calc.rf), hard coral cover (hrd.crl), shoal (loc) and year (Table 3.4). The simplest model (R3) included a single parameter, proportion of hard coral ($R2 = 0.54$) which indicated that fish species richness was higher at sites with greater proportions of hard coral cover (Figure 3.5).

Table 3.4. Top 10 GAMM models examining the effects of year, shoal (loc), hard coral (hrd.crl), plants (plts), bare substrate (bare), boulder (bldr), calcareous reef (calc.rf), gravel (grvl), rubble (rbbl), sand (snd), and aspect (asp) on fish species richness. AICc is the small-sample bias-corrected form of Akaike's information criterion, ΔAICc is the Akaike difference, and w is the Akaike weight. Models with ΔAICc <2 (in bold) are the best fitting models.

M#	model name	AIC$_c$	ΔAIC$_c$	wAIC$_c$	R^2	edf
R1	**calc.rf+loc**	**1473.75**	**0**	**0.23**	**0.55**	**6.97**
R2	**hrd.crl+year**	**1474.26**	**0.51**	**0.18**	**0.55**	**6.32**
R3	**hrd.crl**	**1474.57**	**0.82**	**0.15**	**0.54**	**4.33**
R4	calc.rf	1475.90	2.15	0.08	0.53	5.03
R5	year+hrd.crl.by.year	1475.96	2.21	0.08	0.56	9.59
R6	loc+calc.rf.by.loc	1476.62	2.87	0.05	0.58	9.46
R7	hrd.crl+snd	1477.86	4.12	0.03	0.55	5.38
R8	calc.rf+asp+loc	1477.87	4.13	0.03	0.55	7.97
R9	plants+calc.rf	1478.16	4.42	0.03	0.54	5.94
R10	hrd.crl+snd+year	1478.49	4.75	0.02	0.55	7.31
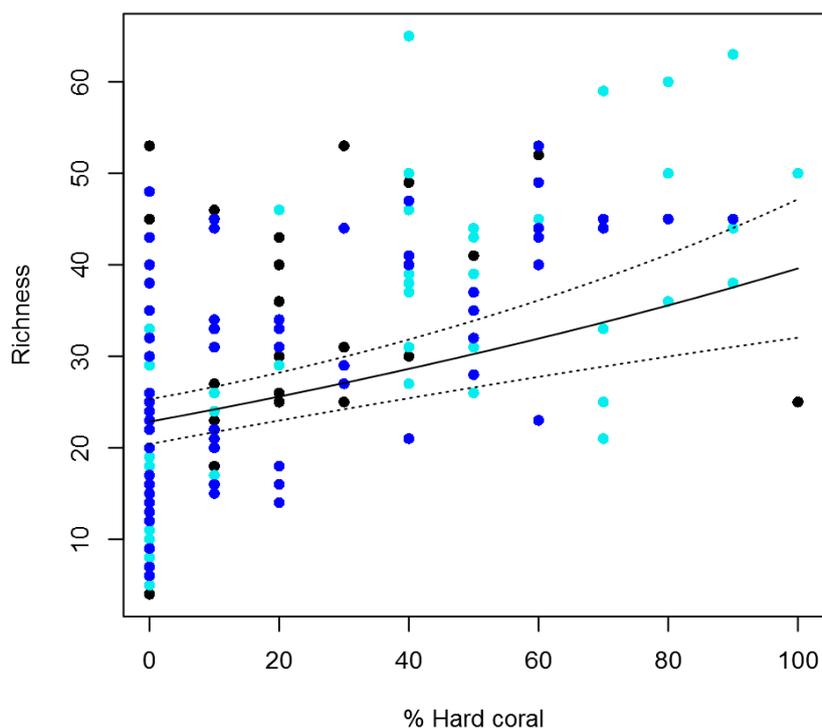
Figure 3.5. Predicted trends in fish species richness with the proportion of hard coral. Solid line indicates the fit of the best-approximating model with 95% confidence intervals (dotted lines) and points coloured by year (black: 2011; light blue: 2013; and dark blue: 2016) .

The best-approximating model for fish relative abundance (A1) included hard coral cover and year (Table 3.5). Model A1 indicated that fish abundance was greater at locations with higher hard coral cover, and during 2011 (black lines, Figure 3.6). The trend in abundance with year followed a pattern in which abundances observed were higher in 2011, lower in 2013, then higher again in 2016 (Figure 3.6). Since the trend in abundance does not increase or decrease consistently with time, and little support for the model where abundance changes as an interaction with year, this variation with year is likely to be unrelated to released hydrocarbons.

Table 3.5. Top 10 GAMM models examining the effects of shoal (loc), year, hard coral (hrd.crl), plants (plts), bare substrate (bare), boulder (bldr), calcareous reef (calc.rf), gravel (grvl), rubble (rbbl), sand (snd), and aspect (asp) on fish abundance. AICc is the small-sample bias-corrected form of Akaike's information criterion, ΔAICc is the Akaike difference, and w is the Akaike weight. Models with ΔAICc <2 (in bold) are the best fitting models.

| M#	model name	AIC_c	ΔAIC_c	$wAIC_c$	r^2	edf
A1	**hrd.crl+year**	**2125.65**	**0**	**0.58**	**0.37**	**6.18**
A2	year+hrd.crl.by.year	2129.30	3.66	0.09	0.38	8.47
A3	hrd.crl+asp+year	2129.35	3.70	0.09	0.37	7.11
A4	hrd.crl+snd+year	2129.45	3.80	0.09	0.37	7.22
A5	bare+rbbl+loc	2131.56	5.91	0.03	0.35	7.28
A6	rbbl+loc	2132.34	6.70	0.02	0.33	6.20
A7	asp+year+hrd.crl.by.year	2132.93	7.29	0.02	0.38	9.57
A8	snd+year+hrd.crl.by.year	2133.28	7.63	0.01	0.38	9.36
A9	hrd.crl+year+snd.by.year	2133.47	7.83	0.01	0.38	9.42
A10	year+hrd.crl.by.year+snd.by.year	2133.80	8.16	0.01	0.39	11.45
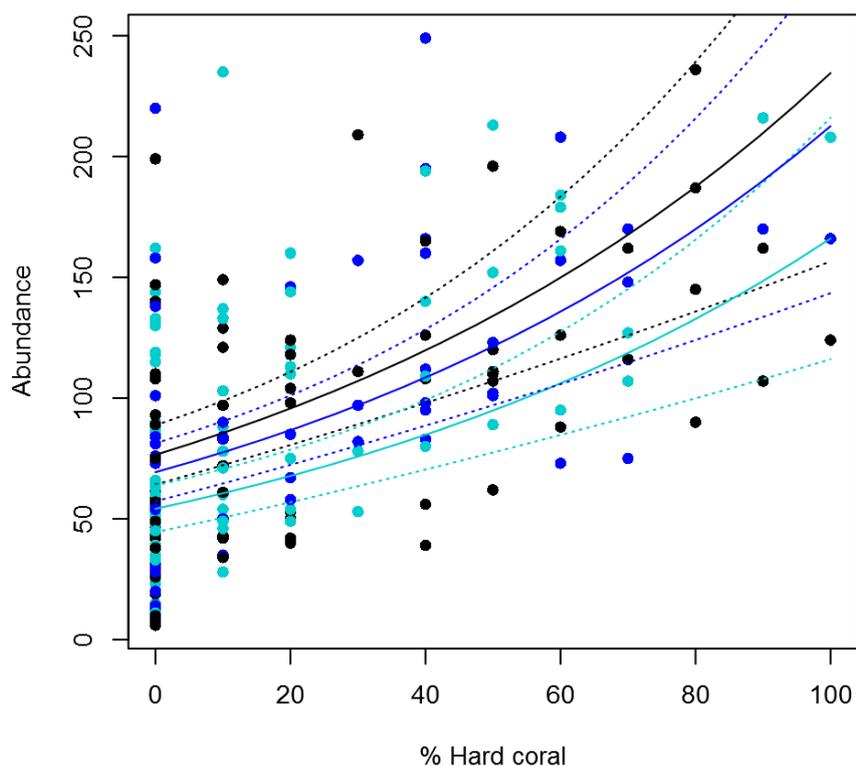
Figure 3.6. Predicted trends in fish relative abundance with proportion of hard coral cover by year. Lines indicate the fit of the best-approximating model coloured by year (black: 2011; light blue: 2013; and dark blue: 2016) with 95% confidence intervals (dotted lines).

3.4.2   Temporal changes in fish communities

A total of 122 species were recorded on at least 5% of the SBRUVS samples at Barracouta East shoal. Redundancy analysis of standardised abundance data for these species showed that the total explainable variation was only 8.29% when the community data was constrained simply by year given site. The effect of year comprised 6.53% and site comprised 1.76% (unconstrained variation 91.71%), and the adjusted R^2 was only 0.36%. The long vector for 2011 in the RDA biplot corresponds with schooling species *Odonus niger*, which implies higher abundance at sites in 2011, than in 2013 and 2016 (Figure 3.7a). Equally, species vectors in the direction opposite the 2011 centroid indicate reduced abundances of species during 2011, such as *Parapercis ciathrata* (Pinguipedidae; sandperch). A number of species were associated with higher abundances in 2013.

At Goeree Shoal, 107 species were observed on at least 5% of samples, and the model explained 8.93% (total variation) of which year comprised 7.02% and site 1.91%. Small schooling Cirrhilabrids were the only prominent species (group) in 2011, with a long species vector that reflects the abundance and grouping of several species that are difficult to identify into one group (Figure 3.7b). The emperor *Lethrinus atkinsoni* were relatively abundant 2011 and 2013. In comparison, multiple vectors were significant for 2016, comprising a number of emperor species (family Lethrinidae), a tilefish *Hoplolatilus cuniculus* (Malancanthidae), a small wrasse (*Pseudochelinus evanidus*) and the *Halichoeres zeylonicus* group, which comprise a number of small abundant fishes. For *Lethrinus atkinsoni* in 2013 and *L. amboinensis* and *L. sp.* in 2016, it is possible the long vectors reflect a change in abundance through time, yet identification of these species is difficult at depth and may represent an artefact of reader effect (e.g. same genus, but perhaps misidentification). For Goeree shoal, the greater number of species highly correlated (*p* < 0.0001; vectors) with the centroids for 2016 could indicate these species which

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occurred at sites that experienced moderate-high exposure to the oil spill were more abundant 7 years since exposure.

Vulcan shoal supported 105 species recorded in over 5% of sites. The model explained 8.65% comprising year (6.63%) and site (2.29%). Trevally (*Caranx melampygus*), emperor (*Lethrinus atkinsoni*) and, with higher numbers of *Lutjanus bohar* (Lutjanidae) were more abundant during the 2011 survey at Vulcan shoal compared to other years (Figure 3.7c). In contrast, 2013 was dominated by smaller-bodied fishes of the *Halichoeres zeylonicus* group and spot-cheek emperor (*Lethrinus rubrioperculatus*) which may be linked to the reduction in seagrass during that year. One species of surgeonfishes (Family Acanthuridae) *Ctenochaetus striatus,* and a schooling triggerfish *Odonus niger* (Balistidae) and sea snake (*Aipysurus laevis*) were most abundant during the 2016 survey.

Overall however, redundancy analyses for year and site did not explain a large amount of variation (Figure 3.7), thus a number of additional factors are likely responsible for the abundance of species in the shoal communities.
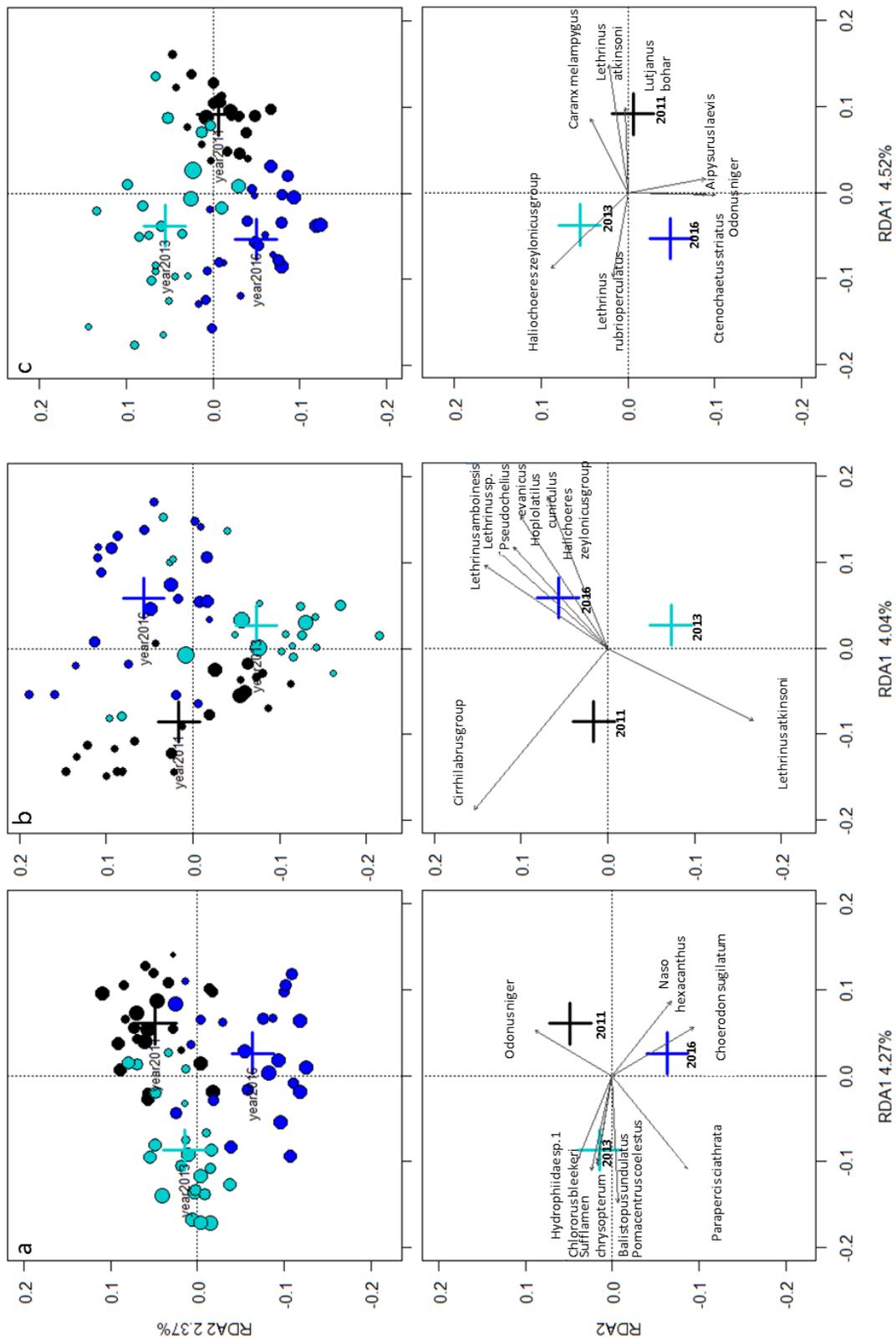
Figure 3.7. Redundancy analysis (RDA) for fish species occurring on at least 5% of samples at each shoal constrained by year and conditioned by site: (a) Barracouta East; (b) Goeree; and (c) Vulcan shoals (n = 122; 107; 105 species). Weighted averages of site scores were scaled by site richness and coloured by year of sampling (2011: black, 2013: light blue; 2016: dark blue). Significant species vectors (p < 0.0001) correlated with linear constraints are shown.

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3.4.3 Temporal and spatial differences in key species length frequency distributions

A total of 2822 measurements were obtained from the six key species across the three shoals over three years. Median lengths of the six key species indicated variation among shoal and year (Table 3.6). Analyses indicated an effect of year and shoal on size for the *Lethrinus* species only. Significant interactions were identified between shoal by year for *L. rubrioperculatus, L. atkinsoni*, and *L. amboinensis,* while for *L. olivaceus*, length varied significantly by shoal and year separately (Table 3.7). Length frequencies for the key species were relatively consistent among shoal and year, with no clear trend observed.

Table 3.6. Summary of the median caudal lengths of the six key species measured by site (B: Barracouta East; G: Goeree; and V: Vulcan shoals) and year, with standard errors (SE).

Shoal year	*Lethrinus rubrioperculatus*		*Lethrinus atkinsoni*		*Lethrinus amboinensis*		*Lethrinus olivaceus*		*Lutjanus bohar*		*Variola albimarginata*	
	Median length (mm)	SE	Median length (mm)	SE	Median length (mm)	SE	Median length (mm)	SE	Median length (mm)	SE	Median length (mm)	SE
B 2011	255.26	3.84	264.84	8.24	317.18	12.56	575.48	40.65	417.22	13.83	272.77	25.12
B 2013	267.44	2.64	304.97	6.15	366.99	17.55	182.65	32.61	421.78	16.91	301.11	49.89
B 2016	268.30	2.62	300.14	6.18	336.93	8.11	608.29	13.68	419.94	10.24	296.38	12.57
G 2011	277.13	4.22	290.21	4.71	358.13	8.82	605.66	46.69	405.38	20.01	330.08	144.46
G 2013	290.88	3.05	306.77	4.64	370.98	7.75	222.90	53.19	455.56	14.68	306.09	43.05
G 2016	281.51	3.09	287.72	3.17	336.73	4.30	562.31	19.29	400.16	21.46	323.27	47.42
V 2011	249.39	7.46	240.65	2.70	362.92	19.60	328.32	52.90	420.29	18.52	309.03	49.74
V 2013	284.67	3.86	244.20	3.30	398.32	34.96	198.09	40.10	361.43	20.52		
V 2016	268.38	5.67	250.83	2.93	366.02	8.26	599.70	20.54	411.20	26.24	280.69	23.27
total *n*	1070		653		386		200		451		64	

Table 3.7. Summary of significant (*p* < 0.05) results from multiple comparison tests of linear models by species. Differences were assessed among shoals (B: Barracouta East; G: Goeree; and V: Vulcan shoals) and year (significant differences: < or >; non-significant: =).

	Lethrinus rubrioperculatus	*Lethrinus atkinsoni*	*Lethrinus amboinensis*
2011	B = G = V	B = G > V	B < G = V
2013	B < G = V	B = G > V	B = G = V
2016	B = G > V	B = G > V	B = G = V
B	2011 = 2013 = 2016	2011 < 2013 = 2016	2011 = 2013 = 2016
G	2011 < 2013 > 2016	2011 < 2013 > 2016	2011 = 2013 > 2016
V	2011 < 2013 > 2016	2011 = 2013 = 2016	2011 = 2013 = 2016

For the most abundant species, *L. rubrioperculatus,* a greater proportion of smaller fish < 200 mm were measured in 2011 (Goeree and Vulcan shoals) and in 2016 at Vulcan shoal, where a clear bimodal distribution was observed with a peak at 150 mm in length (Figure 3.8). Conversely, individuals were larger in 2013 at Goeree and Vulcan shoals.

Figure 3.8. Estimated cumulative density functions (ECDF) plotted over length frequency histograms of the spot-cheeked emperor *Lethrinus rubrioperculatus* **measured on Barracouta East (B), Goeree (G), and Vulcan (V) Shoals, across years (a: 2011; b: 2013; c: 2016. Note the different scales on each of the y-axes.**

At Vulcan shoal, *L. atkinsoni* were smaller in size in comparison to Barracouta East and Goeree shoals with median lengths of ~300 mm. Larger individuals > 350 mm were also more common in 2013 across Barracouta East and Goeree shoals (Figure 3.9b, Table 3.7) compared with other years, while smaller *L. atkinsoni* were observed only at Barracouta East in 2011 and Vulcan shoals during 2011 and 2013.

Figure 3.9. Estimated cumulative density functions (ECDF) plotted over length frequency histograms of the yellowtail emperor *Lethrinus atkinsoni* **measured on Barracouta East (B), Goeree (G), and Vulcan (V) Shoals, across years (a: 2011; b: 2013; c: 2016. Note the different scales on each of the y-axes.**

Of the few small individuals recorded, a larger number of *L. amboinensis* were measured in smaller size classes (< 200 mm) at Barracouta East shoal in 2011 (median length 317 mm) compared to Goeree and Vulcan shoals in the same year (Figure 3.10a). Larger individuals > 400 mm were less common across shoals and years, with more observed at Barracouta East shoal. Within shoals, a decrease in fish size from 2013 to 2016 was observed at Vulcan shoal, yet this shoal comprised fewer individuals measured.
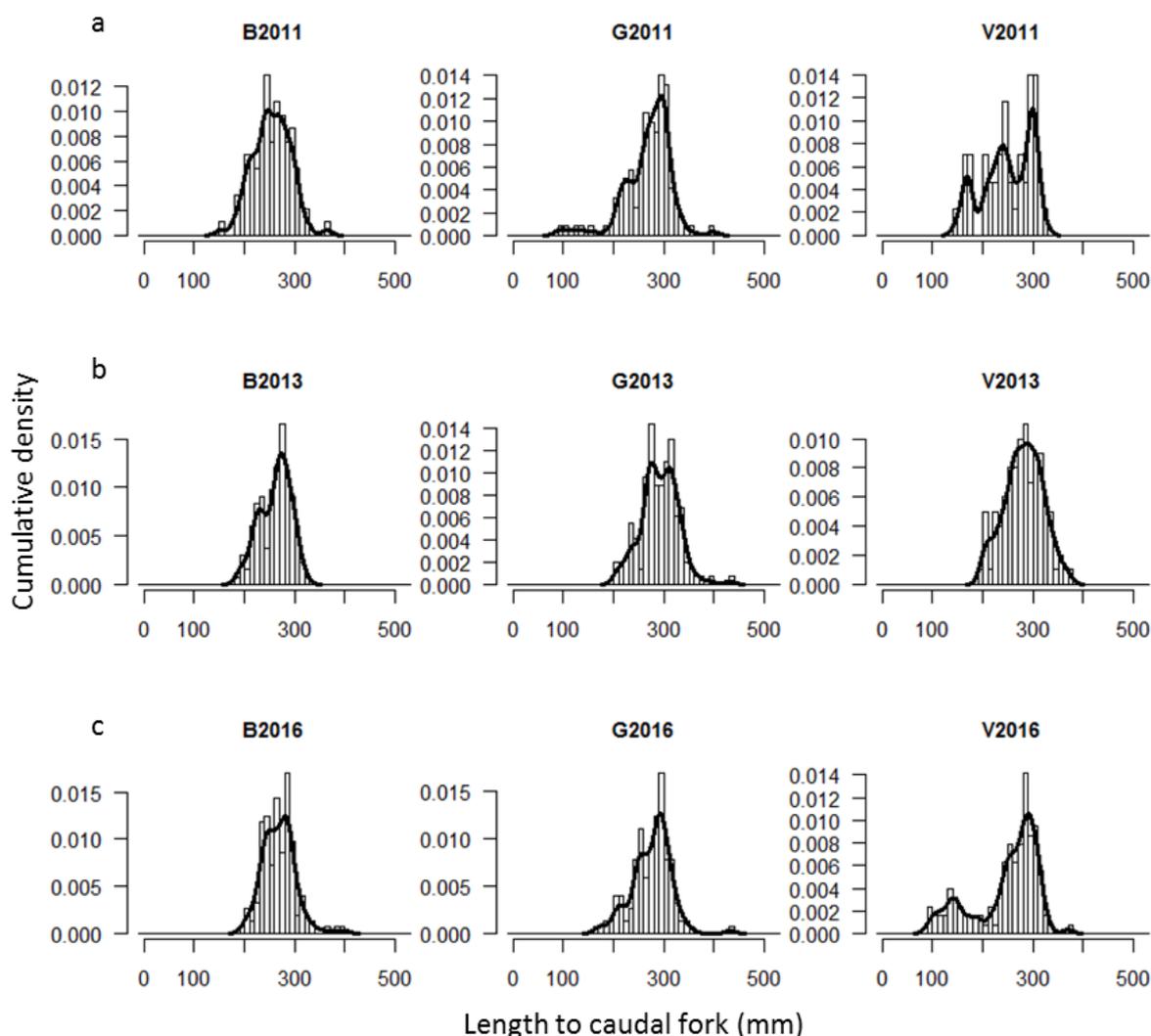
Figure 3.10. Estimated cumulative density functions (ECDF) plotted over length frequency histograms of the ambon emperor *Lethrinus amboinensis* measured on Barracouta East (B), Goeree (G), and Vulcan (V) Shoals, across years (a: 2011; b: 2013; c: 2016). Note the different scales on each of the y-axes.

A bimodal distribution was observed for *L. olivaceus* during 2011 and 2013, in contrast to larger individuals measured at all shoals from the 2016 surveys, with higher median lengths recorded. A high proportion of small *L. olivaceus* individuals (< 150 mm, juveniles) were recorded during the 2013 surveys at all shoals, potentially indicating new recruits (Fig 3.11b).

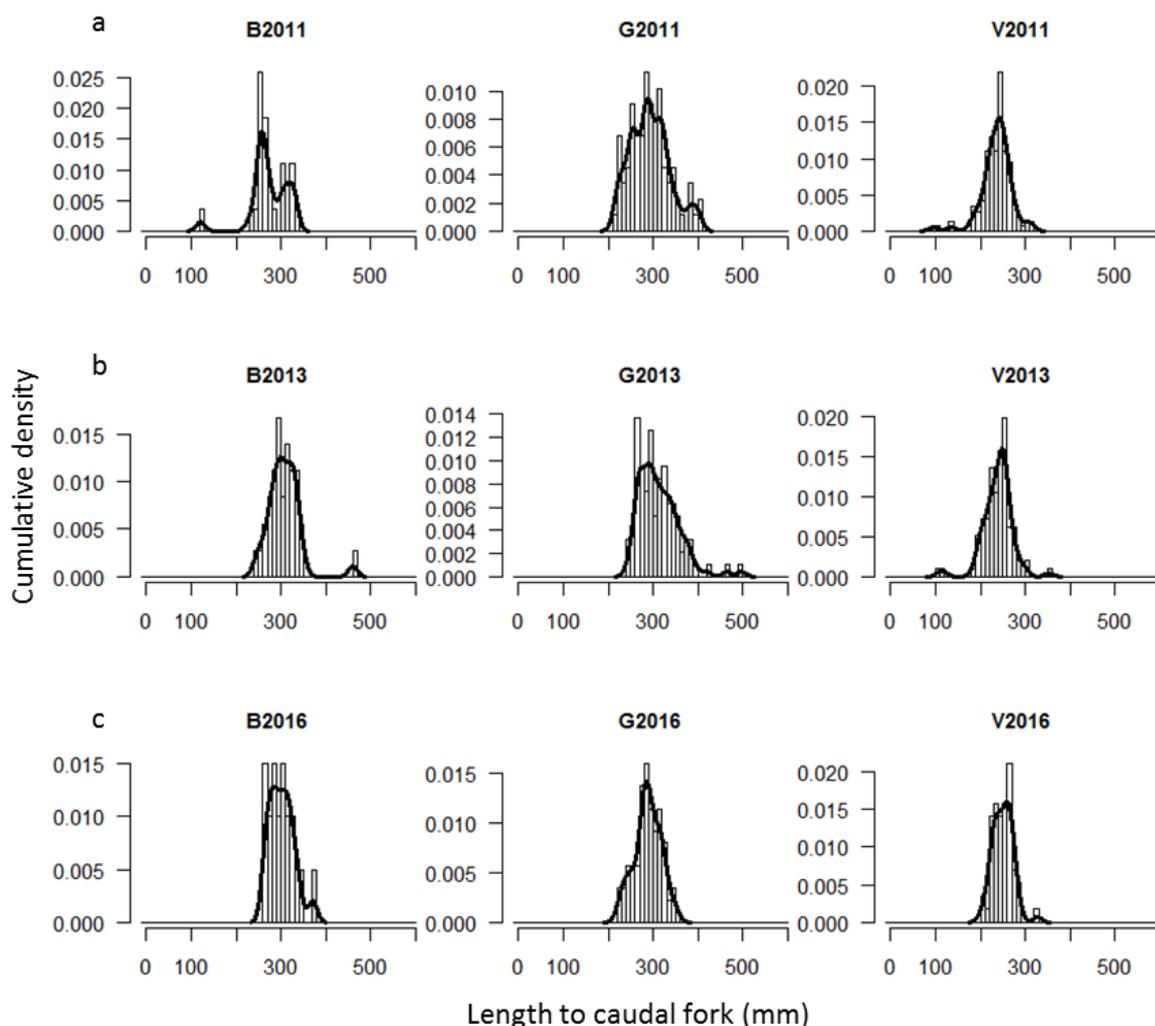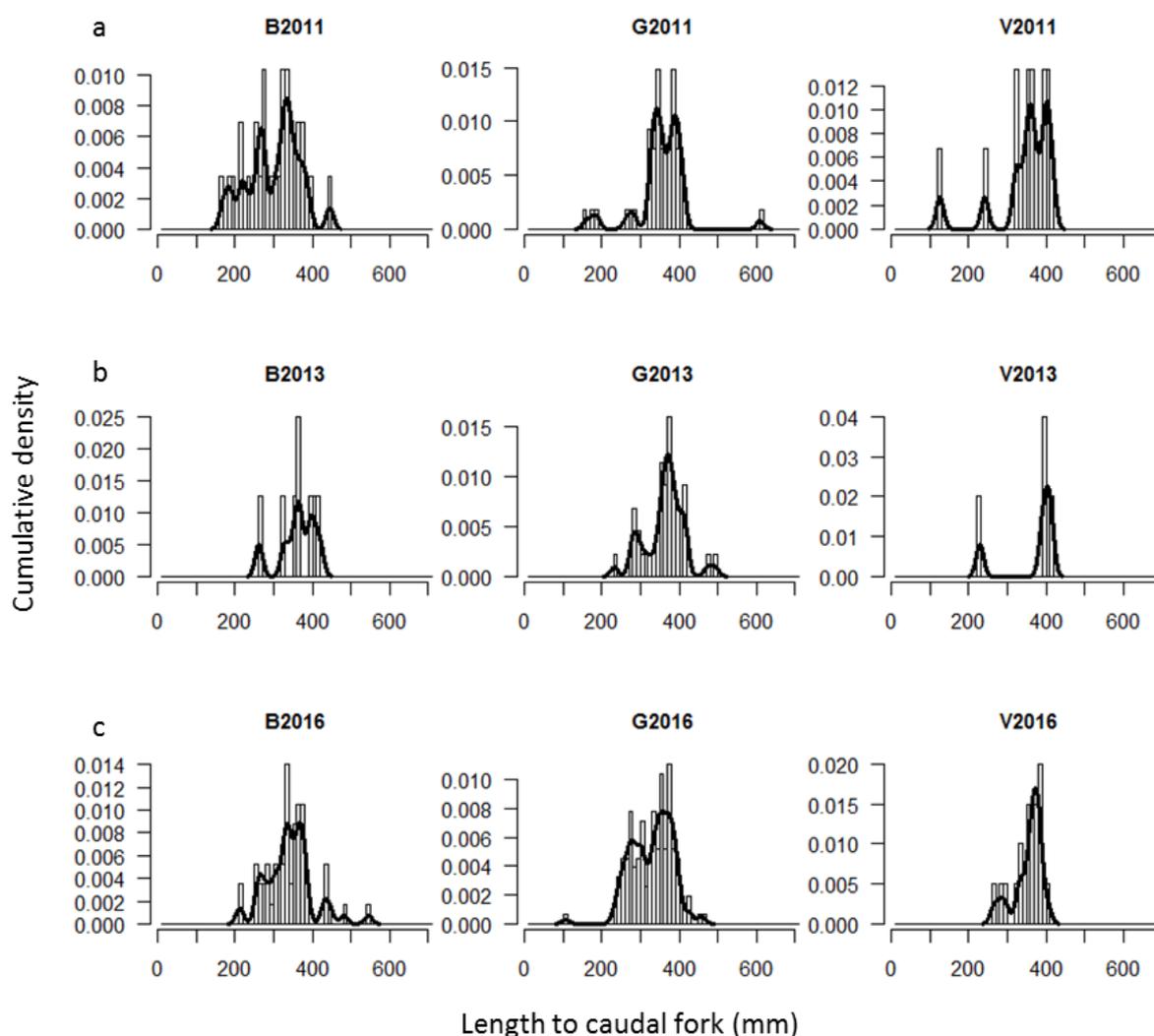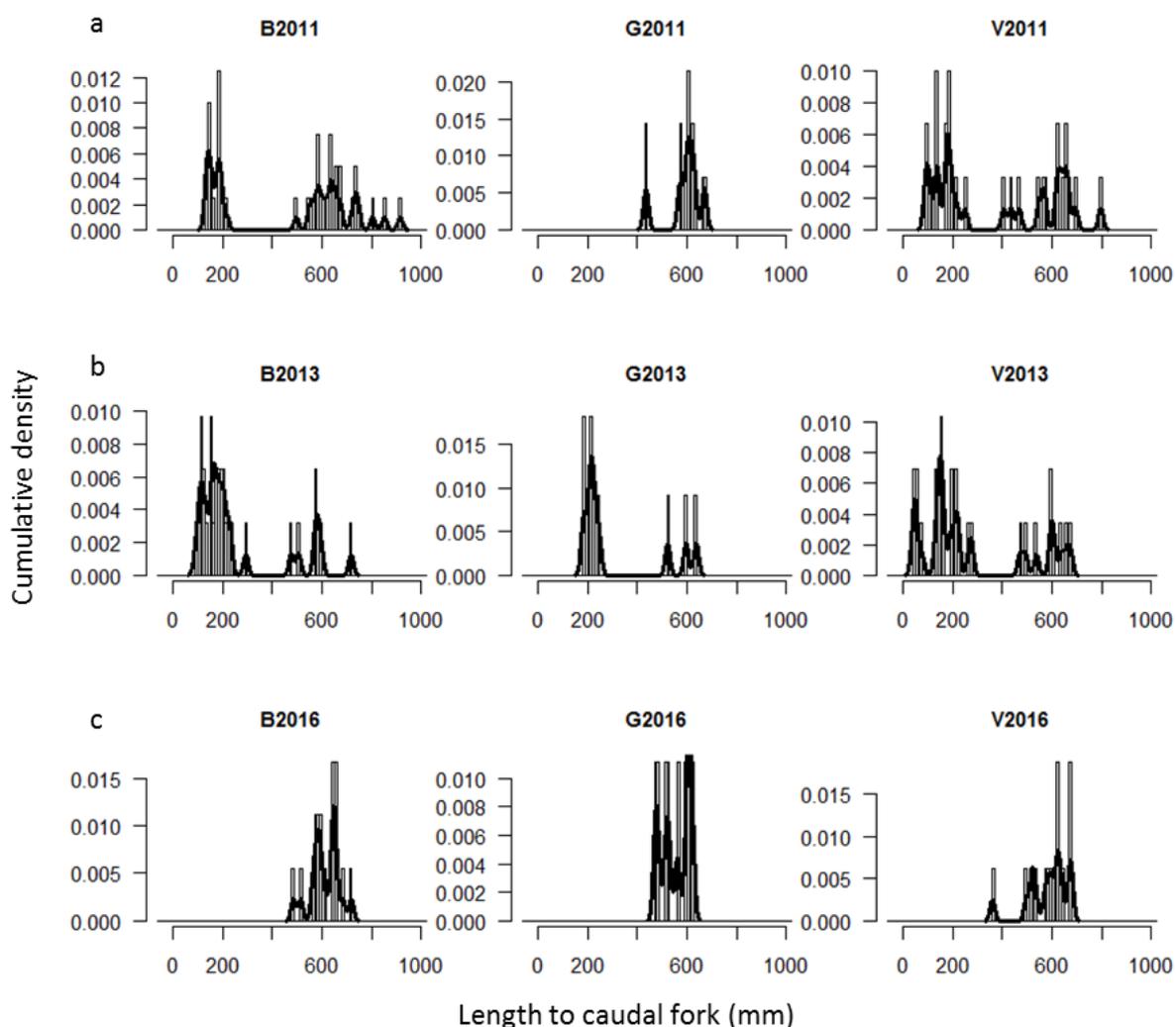**Figure 3.11.** Estimated cumulative density functions (ECDF) plotted over length frequency histograms of the long nose emperor *Lethrinus olivaceus* measured on Barracouta East (B), Goeree (G), and Vulcan (V) Shoals, across years (a: 2011; b: 2013; c: 2016). Note the different scales on each of the y-axes.

A broad size distribution 102.7 mm - 758.7 mm was evident for *Lutjanus bohar* across shoals and years. No significant relationship was observed with time or space, with juveniles and adults common to these three shoals (Figure 3.12).
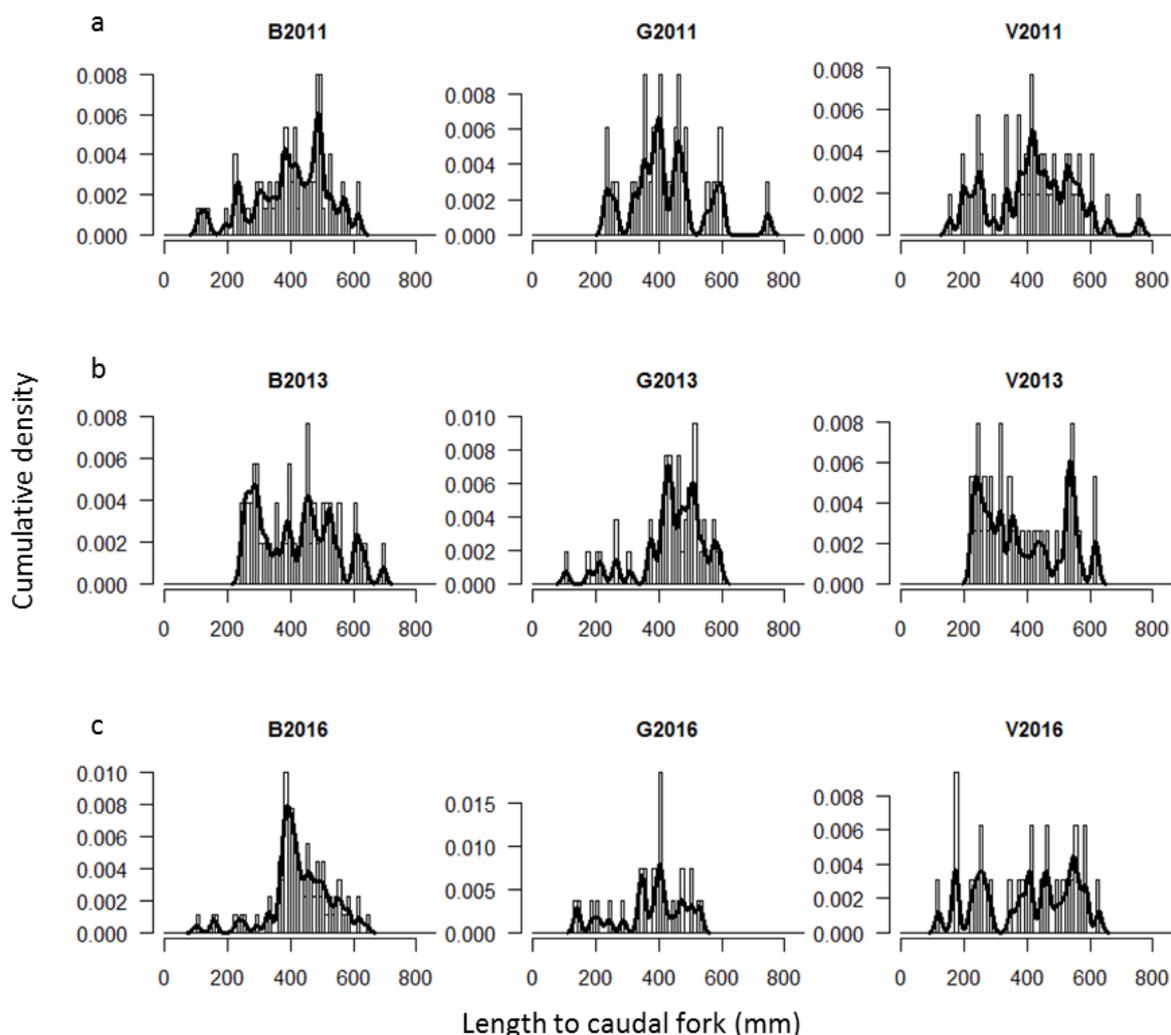
Figure 3.12. Estimated cumulative density functions (ECDF) plotted over length frequency histograms of the red bass *Lutjanus bohar* measured on Barracouta East (B), Goeree (G), and Vulcan (V) Shoals, across years (a: 2011; b: 2013; c: 2016. Note the different scales on each of the y-axes.

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In comparison to the other key species, fewer lengths of *V. albimarginata* were recorded (Figure 3.13). Larger individuals (> 400mm caudal fork length) were recorded at Barracouta East and Vulcan shoals, yet it no effect of year or shoal could be derived from the data.
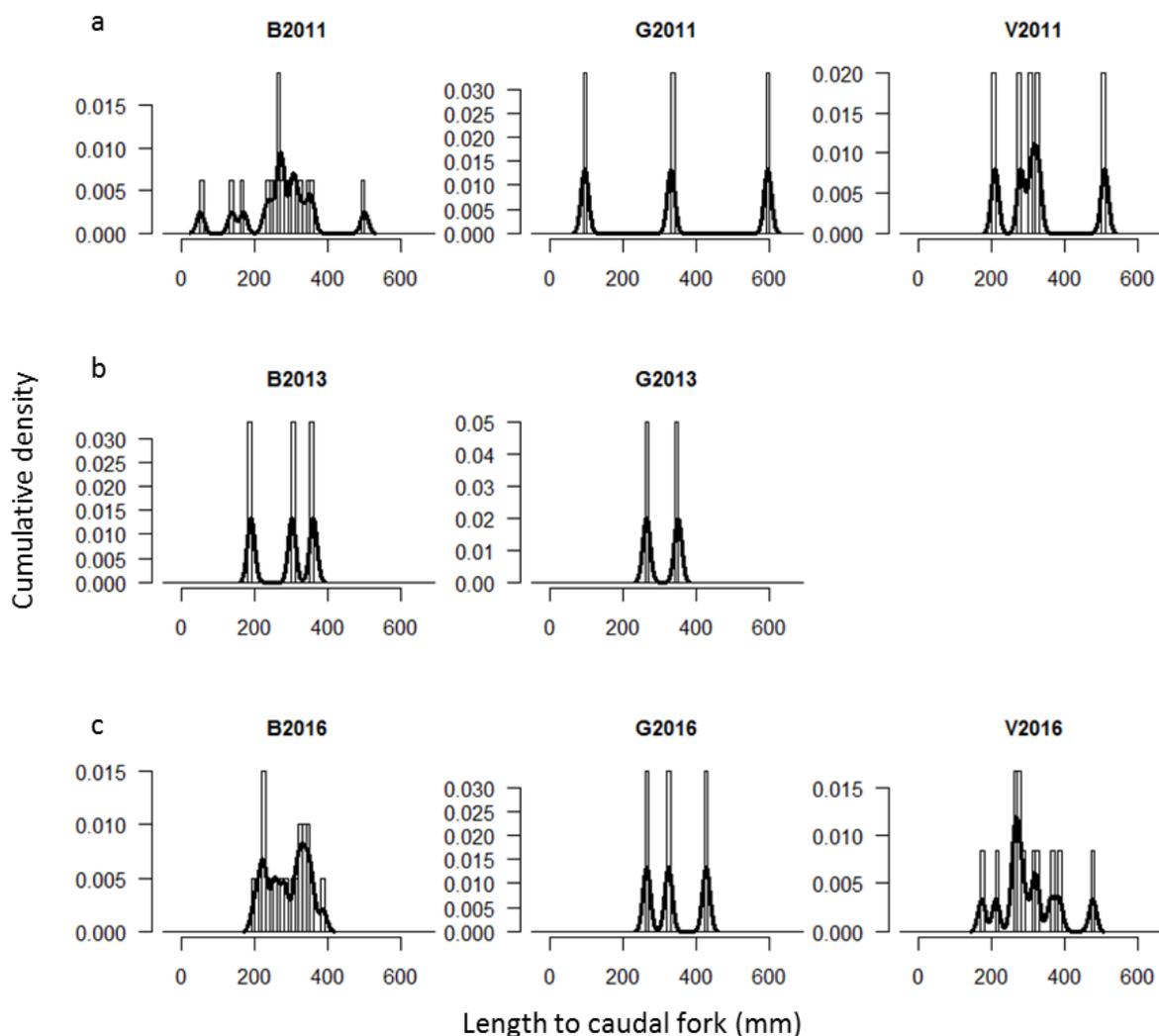
Figure 3.13. Estimated cumulative density functions (ECDF) plotted over length frequency histograms of the white-margined coronation trout *Variola albimarginata* measured on Barracouta East (B), Goeree (G), and Vulcan (V) Shoals, across years (a: 2011; b: 2013; c: 2016. Note the different scales on each of the y-axes.

3.5   Discussion

Monitoring fish communities using BRUVS enables a snapshot of species richness and diversity to be assessed at specified locations that assist in understanding changes through time. Repeat sampling with BRUVs to monitor Barracouta East, Goeree and Vulcan shoals across the three surveys enabled comparison of fish communities through time. To gain an understanding of how the richness and abundance of fishes varied over time and across the shoals, models tested for the effect of year and shoal, abiotic, biotic and spatial variables, with consideration of depth.

3.5.1   Temporal changes in fish richness and abundance

Overall, the variability in richness and abundance of fishes was better explained at the shoal level, rather than among years. This finding is consistent with comparisons of 2011 and 2013 data from the earlier report (Heyward et al. 2013), although some variation among years was observed. Fishes have strong associations with their environment (Roberts and Ormond 1987, McGhee 1994) and their occurrence is related to a combination of physical habitat characteristics (Sale 1991, Friedlander and Parrish 1998, Heyward et al. 2013). Depth has an important role in structuring fish communities, because the abiotic and biotic habitat characteristics that fishes are associated with vary across depth gradients (McGhee 1994, Donaldson 2002, Brokovich et al. 2008, Chatfield et al. 2010). Light levels diminish at depth which may decrease growth rates of algae (Russ 2003) and affect foraging (Rickel and Genin 2005), while gradients in habitat with depth can influence the settlement, growth and survival of recruiting fish larvae (Srinivasan 2003).

Considering the link between depth and fish community composition, richness and abundance of fishes was investigated through time and among shoals with depths categorised as shallow and deep. A shift in species composition at 30 m was apparent for these surveys, with species richness at Barracouta East shoal observed as consistently higher in shallow depths through time. Likewise for abundance, the model estimated higher abundance of individuals at Goeree shoal in depths less than 30 m in 2013. During 2013, all benthic biota decreased in cover except macroalgae, thus, increased abundance of fishes during this survey could reflect herbivorous species or species that are associated with sand habitats. For example, while the RDA for Goeree shoal did not explain a substantial amount of variability, it indicated higher abundance of *L. atkinsoni* in 2013, which has a varied diet of small invertebrates and fishes and forages in sand among coral reef habitat (Carpenter and Allen 1989). The previous report from this study (Heyward et al. 2013) highlighted habitat as a key element in comparisons of richness and total abundance (and size), thus various aspects of the local environment are of greater importance than variation observed among years.

The shoals surveyed differ in location, size, and composition of benthic biota. It is likely that the local fish communities would reflect these differences, and it was not an unexpected result that shoal exhibited a greater effect on richness and abundance than year overall. When richness and abundance were evaluated in relation to year, shoal, biotic and abiotic cover, and aspect (with depth accounted for), both richness and abundance were best explained by percentage hard coral cover. Greater species richness and abundance was related to increased hard coral cover, which is a pattern reported in reef ecosystems (Bell and Galzin 1984, Sano et al. 1984). Thus, decreases in live coral cover, whether due to seasonal fluctuations, bleaching or crown of thorns outbreaks, lead to reduced numbers of coralivores, followed by declines in diversity of other fishes from reduction in habitat complexity (Sano et al. 1984, Coker 2012). The trend in fish abundance and coral cover was also linked to year, with abundance high in 2011, lower in 2013, and moderate in 2016. Lower fish abundance in 2013 was synonymous with decreased benthic cover and increased sand and unconsolidated substrate (rubble) recorded, which was likely linked to the fish species strongly associated with biota. Increased abundance observed during the following 2016 survey could reflect recovery of those benthic-associated fishes in line with the habitat replenishment. While baseline data was not collected at the time of the oil spill, patterns observed are not consistent with expected increase in species richness and abundance post-spill.

Species abundances explained by RDAs constrained by year and conditioned by site did not reveal patterns through time. While Barracouta East shoal indicated a greater number of species had higher abundances in 2013, the same was observed for Goeree shoal in 2016, while abundances for species across families were variable at Vulcan shoal. It is likely that habitat better explains the observed patterns, since benthic cover can change spatially and at small spatial scales and can vary according to positioning of BRUVS on repeated sites from year to year (i.e. placements approximate within 30 m).

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3.5.2 Temporal and spatial effects on fish size

The effect of shoal and year on size was variable among the six key species. While differences in median lengths and size classes were observed during the surveys, there was no consistent pattern detected through time. Likewise, no clear progression of recruits (smaller individuals) to following size classes through the survey years was apparent for the six key species. Since surveys provide a snapshot of fish length composition, repeated sampling across seasons each year would be necessary to identify episodic recruitment across years. Potential recruitment pulses were detected for *L. rubrioperculatus* (2011 at Goeree, 2011 and 2016 at Vulcan shoal) and *L. olivaceus* (2013 all shoals, 2011 Barracouta East and Vulcan shoals), yet further investigation relating length data to habitat variables would provide greater understanding into specific locations the juvenile and adult stages occur. The lutjanid *L. bohar* occurred at sites across all size classes (life stages) at each shoal and year, thus represents a species less likely to be influenced by habitat fluctuations through time.

3.5.3 Summary

In summary, fish communities appear relatively stable in terms of richness and abundance across Barracouta East, Goeree and Vulcan shoals. The three shoals varied in potential oil spill exposure using a Euclidean distance based metric from low (Barracouta East) to high, (Goeree and Vulcan), with Goeree Shoal experiencing slightly lower exposure than Vulcan Shoal (Table 3.1, Heyward et al. 2013). Previous observations indicated that on a shoal scale, all sites had similar habitats characterised by a mix of abiotic sand or gravel and calcareous reef; areas that were not bare were dominated by encrusting organisms and massive and branching corals (Heyward et al. 2012). The effects of hard coral cover, depth and shoal appear to structure fish communities to a greater degree than the changes observed through time.

3.6   Recommendations

This study confirms that the shoals support a persistent period 2010-2016 and highly diverse fish communities, particularly associated with coral reef areas. Strong coupling is present between fish diversity and reef coral communities. As communities, such hard corals have shown more moderate changes, declining then increasing over the study period, fish diversity has responded accordingly. Observed temporal variability in fish diversity and abundance has been very stable as the whole of shoal scale and unrelated to distance from the MWHP.

The shoals support a range of fish communities that may respond differently to both natural and human pressures. Broad-scale monitoring using BRUVS has proved effective tool for monitoring these fish communities. Research is underway to automate as much BRUVS image analysis and when implemented it will provide a tool for rapid overview of the status of key fish families in different habitats each shoal. In combination with rapid image classification, via tools such as BenthoBot, general monitoring in the future will become more cost effective.

As a result adaptive monitoring of shoal habitats is likely on the near horizon. Even at its present level of development Benthobot and analysis of BRUVS images, if provided with additional field data collection, would be capable of detecting a major departure in the abundance of shoal fish and habitats from natural perturbated levels and in turn trigger a more resource intensive assessment if required.

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