Developing indicators and a baseline for monitoring demersal fish in data-poor, offshore Marine Parks using probabilistic sampling

Authors

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Abstract

The number of Marine Protected Areas (MPAs) has increased globally as concerns over the impact that human activities are having on the world's oceans have also increased. Monitoring is a key requirement to determine if MPAs are meeting their objectives. However, many recently declared MPA's are large, offshore, or form part of an expansive network and spatial information about the habitats, communities and species that they contain is often lacking. This presents challenges for deciding exactly *what* to monitor and developing strategies on *how* to monitor it efficiently. Here we examine these issues using the Flinders Marine Park in Australia as a case study. We trial a two-stage version of a spatially-balanced, probabilistic sampling design combined with Baited Remote Underwater Videos (BRUVs) to perform an initial inventory, and we evaluate the potential of six commercially and ecologically important demersal fish as indicators within the Marine Park. Using this approach we were able to (1) quantitatively describe the distribution of the fish species in the Marine Park; (2) establish quantitative and representative estimates of their abundance throughout the Marine Park to serve as a baseline for future monitoring; (3) conduct power analyses to estimate the magnitude of increase we may be able to detect with feasible levels of sampling effort. Power analysis suggested that for most of our potential indicator species, detecting increases in abundance as small as 50% from present values should be feasible if sampling is restricted to a species' preferred habitat and the same sites are sampled through time. Our approach is transferrable to other regions where monitoring programs must be designed based on limited spatial and biological data, assisting with decisions on *what* and *how* to monitor.

Keywords: Baited Remote Underwater Videos (BRUV), Marine Protected Area (MPA), spatially-balanced sampling, Generalised Random Tessellation Stratified (GRTS) sampling.

1. Introduction

Spatial management options are becoming increasingly prevalent as concerns escalate over the impact that humans are having on marine ecosystems. These impacts include declines in key species, loss of biodiversity (Worm et al., 2006), catastrophic ecosystem regime shifts (Johnson et al., 2011), and climate-related range shifts (Poloczanska et al., 2013). An important tool in the conservation toolbox is Marine Protected Areas (MPAs) which are regions of ocean afforded varying levels of protection from human interference. Currently, MPAs cover approximately 7% of the world's ocean (UNEP-WCMC and IUCN, 2017). Recent MPAs tend to cover large areas (Spalding et al., 2013) or incorporate a series of interconnected MPAs, a strategy that is generally more effective at achieving conservation objectives (Edgar et al., 2014). However, in order to assess whether MPAs are meeting their objectives, and to inform adaptive management, carefully designed monitoring programs that track changes in the abundance and/or health of indicator species, key groups or assemblages are required (Ferraro and Pressey, 2015). In contrast to the numerous studies that report on monitoring programs and the effects of MPAs in coastal waters (e.g. Barrett et al. (2007), Denny et al. (2004), Kelaher et al. (2014)), fewer studies exist for large, and often remote, offshore MPAs (but see Alemany et al. (2013)). Developing monitoring programs for offshore MPAs is difficult. Often there is a lack of baseline data or detailed prior knowledge on the distribution of habitats and ecological features upon which to build a monitoring program that is inherently spatial. Combined with the logistics of working in remote environments, this presents challenges for deciding exactly *what* to monitor and developing strategies on *how* to monitor it efficiently.

Here we examine these issues in the Australian context. In 2012, the Australian government proclaimed a network of Australian Marine Parks (DOTE, 2014). The network amalgamated 33 previously declared Marine Parks with 27 new Marine Parks. Protection within the Marine Parks ranges from sanctuary zones (IUCN 1a) to multiple use zones (IUCN VI). The network covers approximately 3.1 million square kilometres, all of which is offshore (>3 nm), a large proportion of which covers deep waters (> 100 m), and some of which is remote and difficult to access. The network of Marine Parks aims to 'reasonably reflect the biotic diversity of marine ecosystems' (Althaus et al., 2017; DOTE, 2014). Following the declaration of the network, there is a need to develop monitoring programs to evaluate the performance of individual Marine Parks. Whilst the best available information was used to delineate the Marine Parks and their values, the objectives of specific Marine Parks are often quite broad including, for example, protecting habitats, communities and ecosystems representative of the region (Director of National Parks, 2013). Translating these high level objectives into tangible metrics for monitoring can be difficult. This is exacerbated by the fact that while broad-scale biogeographic information was available to delineate the Marine Parks, fine-scale, spatially explicit information on benthic (and pelagic) habitats as well the composition and distribution of communities and key species is often lacking (Lawrence et al., 2015). As a result, monitoring programs must begin with an inventory to inform *what* exactly to monitor. In addition, the vast size of Marine Parks and their remoteness means larger vessels must be used which increases costs, there are large distances between sampling sites, sites may not be able to be sampled due to weather or other unforeseen circumstances, multiple sampling gears will be used concurrently to satisfy multiple objectives and sampling should be non-destructive. This has implications for *how* to monitor.

Choosing appropriate indicators for monitoring complex biological systems can be difficult, even when prior knowledge of the management region is good; a difficulty illustrated by the profuse number of ecological indicators proposed for marine systems (Teixeira et al., 2016). Here we consider demersal fish species as potential indicators in a long-term monitoring program within the Marine Park network. Demersal fish are often a significant component of the biodiversity that MPAs are intended to protect. They typically have smaller ranges than pelagic species and hence are more likely to be responsive to management interventions, such as zoning arrangements, within MPAs. In qualitative modelling undertaken during the development of Australia's Marine Park network, demersal fish emerged as consistent and sensitive indicators on the state of a range shelf ecosystems (Dambacher et al. , 2011). Demersal fish have proven useful indicators of the effectiveness of smaller and/or more coastal MPAs (Barrett et al., 2007; Bornt et al., 2015; Denny et al., 2004; Stuart-Smith et al., 2017) and they are also relevant to and easily interpreted by management and the public. Thus demersal fish fulfil several key criteria for selecting indicators as recently summarised by Hayes et al. (2015). However, indicators must also be feasible to monitor and this is influenced by their abundance, distribution and variability across the region, as well as the availability of suitable monitoring equipment, and we assess these aspects using relatively inexpensive sampling methods for six key demersal fish species.

Choosing an appropriate sampling design and sampling gear are a core component of *how to* monitor. Sampling designs used for inventory and monitoring in large, offshore Marine Parks must be able to draw inference across the whole region of interest with relatively few sites. They must also be sufficiently flexible to accommodate multiple and potentially changing objectives. Sampling that is representative of a region is best achieved using a probabilistic sampling design, where every part of the sampling region has a quantifiable probability of being selected (Smith et al., 2017). This includes randomised designs but contrasts with judgemental sampling, where sites are chosen *a priori* based on expert knowledge or some other criteria, that is sometimes used for monitoring coastal MPAs (e.g. (Barrett et al., 2007). The best known probabilistic sampling design is simple random sampling (Thompson, 2012). Whilst simple random sampling provide unbiased estimates of the status and trends within an MPA, they may not be *efficient* in that many sites may be required to reduce uncertainty to acceptable levels. An emerging alternative is to utilise spatially-balanced probabilistic designs (e.g. Robertson et al. (2013); Stevens and Olsen (2004)) and here we trial and evaluate the use of one such design called Generalised Random Tessellation Stratified (GRTS) sampling (Stevens and Olsen, 2004). GRTS is flexible and efficient at achieving spatially-balanced sampling under a range of scenarios, and has shown promise for monitoring natural resources in aquatic systems (e.g. Dambacher et al. (2009)).

Sampling gear is another key consideration for monitoring, and within MPAs the choice of sampling gear is ideally non-extractive. However, the depth of the Australian Marine Park network precludes the use of some traditional non-extractive approaches such as SCUBA-based surveys. Baited remote underwater stereo videos (stereo BRUVs) have been effective in censuses of fish in coastal waters (Malcolm et al., 2007; Watson et al., 2009) and here we examine their use for monitoring within the Australian Marine Park network.

We focus on demersal fish within the Flinders Marine Park (FMP) shelf, which is one of the Marine Parks within the Australian Marine Parks network, as a case study for determining *what* and *how* to monitor in a region where little prior knowledge is available. The Flinders Marine Park was established in 2007 and the continental shelf is a multiple use zone (IUCN VI). Knowledge on the spatial distribution and abundance of benthic habitats, communities and key species in this region is very limited and our work forms part of a broader survey program conducted in 2012 aiming to redress this issue to inform monitoring efforts (Hill et al., 2014; Lawrence et al., 2015; Monk et al., 2016). We trial and evaluate the efficacy of using the GRTS spatially balanced survey design, using BRUVs, for developing representative baseline estimates of the distribution, size structure and relative abundance of six species of demersal fish within the Flinders CMR shelf (i.e. the *how* to monitor). To evaluate *what* to monitor, we conduct power analyses using GRTS -based estimates to determine the magnitude of trends that we may be able to detect for each of the six species with feasible sampling effort.

2. Materials and methods

2.1. Study Region

The Flinders Marine Park (FMP) lies offshore of the north-east of Tasmania extending from approximately 35 m depth to 3,000 m depth. Our study region is restricted to the ~813 km^2 multiple use zone on the continental shelf between 40 and 180 m (Figure 1); where most of the anthropogenic pressure is concentrated. The region falls within the Commonwealth Fisheries South East Trawl Sector and the Gillnet and Shark Hook Sector. Low to moderate (~2,000 kg/year) commercial fishing effort occurred on the shelf before the declaration of the FMP in 2007. Demersal trawling was concentrated on the outer shelf, while hook, line and gillnet fishing were more dispersed across the shelf (Pitcher et al., 2016). Since 2007 demersal trawling has been prohibited (Director of National Parks, 2013). At the same time, the Australian Fisheries Management Authority (AFMA) imposed a ban on hook and line methods for the area overlaying the FMP as part of an AFMA closure to protect Harrisson's Dogfish (*Centrophorus harrissoni)*. Reassessment of this closure in 2013 resulted in the shelf (< 180 m) being re-opened to hook and line methods (AFMA, 2012; Williams et al., 2013). Gillnets and recreational fishing are also allowed on the shelf of the Marine Park.

The FMP was established to protect 'representative examples of the ecosystems, communities and habitats' associated with the Tasmanian Shelf and Southeast Shelf Transition biogeographic provinces (Commonwealth of Australia, 2014; Director of National Parks, 2013). These provinces are considered cool-temperate in climate and are influenced by the east Australian current which brings warmer waters onto the shelf during summer (Harris et al., 1987). The FMP experiences high, but variable, wave exposure and tidal currents (Fandry, 1983). On the shelf, benthic habitats consist of large swaths of sediment interspersed with low-profile and sediment-inundated rocky reefs. Steep rocky outcrops occur at the shelf break and heads of canyons (Lawrence et al. 2015). Rocky reefs support dense aggregations of high profile sponges in the shallow sections (<70 m) of the FMP shelf and lower-profile sponges and byrozoans in deeper sections (Lawrence et al. 2015, Monk et al. 2016). In turn, reefs support distinct and more diverse fish assemblages than sediments in the region, although fish diversity decreases with depth on both habitat types (Hill et al. 2014).

2.2. Sampling design and collection methods

Fish species were sampled with stereo BRUVs as part of a larger survey program undertaken in August 2012 that also aimed to inventory and baseline habitat types and reef-associated macro-invertebrate assemblages within the FMP. Because detailed knowledge of the spatial distribution of habitats types on the FMP shelf was limited prior to our survey, a two-phase survey was conducted. The first phase was concerned with quantifying substrate from which habitat types were inferred (described in Lawrence et al. (2015)) across the FMP shelf and in the second phase a subset of sites were revisited to describe and quantify biological assemblages. In both phases the probabilistic design, Generalised Random –Tessellation Stratified sampling or GRTS, was used to select sampling sites.

An ordered master list (Larsen et al., 2008) of all possible GRTS sites was generated. In the first phase of the survey the first 40 GRTS sites were visited and classified as either ‘soft’, sediment habitat or ‘mixed’ low-profile reef and sediment habitat as described in Lawrence et al. (2015) and shown in Figure 1. Because of the 1 hr soak time of the stereo BRUVs and the large size of the survey area, we were unable to revisit all phase one sites within the time available. Instead, in phase two of the survey we used a 2-stage design for sampling with BRUVs. Here a subset of phase one sites were revisited as well as clusters of sites surrounding each of the phase one sites that were selected from the GRTS master list. The subset of sites sampled mixed reef habitats more intensively than sediment habitats because shelf reef systems within the Marine Park are recognised as an important conservation feature (Director of National Parks, 2013). The first eight 'mixed' reef and three 'soft' sediment sites from phase one were sampled with stereo BRUVs. A judgemental site (i.e. chosen using expert opinion) at the head of a shelf-incising canyon was also sampled. In the second stage of the BRUVs sampling, a cluster of four sites within 1km of each revisited GRTS (and the canyon head) site was selected sequentially from the GRTS master-list (see Hill et al. (2014) for further details). There were a few exceptions to this. Four of the second stage BRUV sites were judgementally chosen to ensure they fell on reef features identified using multibeam and replaced the last GRTS site in that cluster. Since any spatial sub-sample of a GRTS master list is also a spatially balanced GRTS sample (Larsen et al., 2008), as long as the sub-sample maintains the same order as the master list, our 2nd stage sampling design is also spatially balanced and can be treated like a standard 2-stage GRTS sampling design. From a logistical perspective, deploying clusters of BRUVs in this manner meant that we were able to deploy and retrieve a cluster of five BRUVs in approximately two hours before steaming to the next cluster which was substantially more efficient than completing one BRUV deployment in approximately one hour before a comparable steam to another phase one GRTS site.

The stereo BRUVs used for sampling consisted of two Canon Legria HFM-300 digital camcorders fitted with Raynox 50 mm wide angle lenses. Cameras were mounted in PVC housings on a weighted galvanized steel frame 700 mm apart angled inwards at 8 ° and approximately 500 mm off the ground. A synchronizing diode arm with mesh bait bag attached extended 1200 mm in front of the cameras. BRUVs deployments used protocols that are standard for coastal regions of temperate Australia (Harvey et al. 2012). One kilogram of crushed pilchards (*Sardinops neopilchardus*) was used as the bait attractant. Adjacent, concurrent drops were separated by at least 250 m to avoid overlap of bait plumes and reduce the likelihood of fish moving between sites and the sampling period was 60 min. Seven Royal Blue CREE XLamps XP-E LEDs (delivering a radiant flux of 350-425 mW at wavelength ranging from 450 to 465 nm) were used to illuminate the stereo BRUVs field of view in depths greater than 60 m (Fitzpatrick et al. 2013). BRUVs were deployed during daylight hours.

2.3. Potential indicator species

Six species were chosen for investigation as potential indicators based on their prevalence in the FMP BRUV footage (see Hill et al. (2014)) and their ecological and fisheries relevance. The species we consider are: *Helicolenus percoides* (Reef ocean perch), *Latris lineata* (Striped trumpeter), *Mustelus antarcticus* (Gummy shark), *Nemodactylus macropertus* (Jackass morwong), *Platycephalus richarsoni* (Tiger flathead) and *Platycephalus bassensis* (Sand flathead). Many of these species have wide distributions spanning temperate Australia and undergo ontogenetic shifts in habitat use. They are targeted by both commercial and recreational fishers using various fishing gears (some of which are prohibited in the multiple use zone). Information on the distribution and ecology of each species and associated fisheries is summarised in Table 1.

2.4. Video scoring

Stereo BRUV pairs were calibrated following procedures outlined in Harvey and Shortis (1995) using the CAL software (www.seagis.com.au). The relative abundance of fishes were estimated using maximum number of fish occurring in any one frame for each species (MaxN; Ellis and Demartini (1995)). Fish within a standardised 5 m field of view of the bait bag were scored. The length of the six indicator fish species were recorded for as many individuals as possible occurring within frames adjacent to MaxN as some individuals were obscured by other fish. Scoring and measuring were completed in the software Event Measure (www.seagis.com.au).

2.5. Distribution of species across the Marine Park

We describe the distribution of candidate indicator species across the Flinders Marine Park with respect to three readily available environmental variables; the substratum type determined in phase one of the sampling program, depth, and latitude. The abundance of each species was modelled against predictor variables using Generalised Linear Models (GLMs) with a negative binomial error distribution. The exception was *M. antarcticus* which was converted to presence-absence data (there was only one site with > 1 individual recorded) and modelled using a binomial error distribution. Parameter estimates and their standard errors were generated as the mean and standard deviation of 10,000 Bayesian bootstrap samples (Rubin, 1981). Predictor variables were considered to be significant if estimates of the coefficient's 95% confidence interval did not overlap with zero. The raw data for each species contained a high proportion of zeros and model fit was assessed by the ability of the models to predict the proportion of zeros observed as well as the average deviance explained. All statistical analyses were conducted in R (R Development Core Team, 2015).

2.6. Patterns in the size of species

The length distributions of individuals at MaxN were qualitatively examined with reference to Tasmanian recreational fishery size limits for each species using histograms. The length of fish at MaxN was modelled against environmental variables using linear models for *M. antarcticus, P. bassensis,* and *P. richardsoni* where only 1-2 individuals were recorded or could be measured at MaxN for the large majority of deployments. The remaining species were modelled using linear mixed effect models where each BRUV site was considered a random effect (Bates et al., 2015). This allows correlation to be induced within the site – fish of similar size are likely to, by stochastic chance, be found together. Coefficients describing the effect of the environmental variables were estimated using 10,000 parametric bootstrap samples and restricted maximum likelihood (REML). As above, environmental variables were considered to be significant if the 95% confidence interval of the coefficient estimate did not overlap with zero.

2.7. Estimating the status of species within the Marine Park

We used design-based estimates to quantify the relative abundance (and therefore current status) of each species within the FMP. The design-based estimates scale the relative abundances of a species recorded at each deployment to an estimate for the entire Marine Park, taking into account how we have sampled. This park-level estimate can be used as a present-day baseline against which future relative abundances can be measured and park performance evaluated. We note that in some circumstances alternative baselines may be more appropriate for measuring performance, such as a return to pre- pressure conditions, however robustly estimating these is not trivial. Obtaining a park-level estimate is possible because we used a probabilistic sampling design as opposed to a judgemental design where sites are chosen on based on *a priori* knowledge or some other factor and are therefore not statistically representative.

We estimated the mean MaxN (as well as the standard error and 95% confidence interval of the mean) for each species across the entire FMP shelf and in each substrate type within the FMP separately using the *total.est* function in the *spsurvey* package by Kincaid and Olsen (2015). The few judgemental sites we sampled were excluded in design-based estimates, resulting in 30 mixed reef and 12 sand GRTS sites with responses. The probability of inclusion was unequal due to the survey design, and so sites were weighted based on the inverse of their inclusion probability. Inclusion probabilities were calculated for both stages. Inclusion probabilities for stage-1 sites were calculated separately for mixed reef and soft sediment substrate and take in to account the overall proportion of each substrate, established from the initial 40 sites sampled for substrate type in phase one (Lawrence et al., 2015). Inclusion probabilities for stage-2 sites were calculated using the number of sites within 1km of the selected phase one site. For estimates across both substrate types in the FMP, the strata feature within *total.est* was used to account for the oversampling of mixed reef strata for the BRUV drops. The *total.est* function was also used to calculate the variance of the mean estimator. This function takes the two-stage sampling and the uneven inclusion probabilities into account when calculating variances. We specified nearest neighbour variance estimation, but where there were less than four sites in any strata or cluster, the naïve variance estimate that assumes independent random sampling, was used as the nearest neighbour estimator is not available in these situations (Kincaid and Olsen (2015).

2.8. Power analysis

Estimating the level of sampling needed to detect trends of various magnitudes is useful for assessing the feasibility of monitoring the candidate indicators. Ideally, some time series data would be available to aid in the estimation of temporal effects. Since we only have one sampling event we conducted power analyses with simplistic assumptions to gain a coarse estimate of feasibility. We expect that the abundance of demersal fish would increase under the removal of fishing pressure. Therefore, we determined the approximate number of BRUV deployments required at each sampling event to detect a 50, 100 and 200 percent increase in mean relative fish abundance (mean MaxN) between two sampling events within the FMP for scenarios where; (1) the same sites are revisited (i.e. a paired *t*-test), and (2) new sites are sampled (i.e. an un-paired *t*-test). Here each site is a BRUV deployment, with the configuration of sites following the same 2-stage clustered design as used throughout the study. The significance level for detecting a difference between the sampling events was set at 0.05, and the power to detect an effect set at 0.8. The effect sizes corresponding to a 50, 100 and 200% increase in MaxN were calculated using Cohens-D formula (which is essentially the standardised mean difference between MaxN at the two sampling times (Cohen, 1988)) using design-based estimates of mean MaxN for each species and an appropriate multiplier for sampling event 2 (i.e. 1.5 for 50% increase and so on). The pooled variance used the design-based estimate of variance at sampling time 1 as we have no other information available to estimate temporal variance and no reason to assume that variance will change between the two sampling events. Since we are interested in detecting an increase in MaxN, tests were one-tailed. Separate power calculations were run for each species and for each habitat (i.e. all habitats combined, mixed and sand substrata). An additional power analysis was run on the mean MaxN of large-bodied fish (> 250 mm) of all indicator species combined as large-bodied fish have proven effective indicators of MPA effects in previous studies (Bornt et al., 2015; Stuart-Smith et al., 2017). Power analyses were carried out using the R statistical package “pwr” (Champely, 2007).

3. Results

3.1. Patterns in distribution and size of species

A total of 51 stereo BRUVs sites were successfully deployed for this study, of these 42 were GRTS sites. Of the potential indicator species, *Nemadactylus macropterus* was the most abundant with 379 individuals recorded at MaxN across the 51 deployments. *Latris lineata* was the next most abundant with 95 individuals recorded at MaxN across all deployments, followed by *Helicolenus percoides* (65 individuals). *Platycephalus bassensis* and *P. richardsoni* were relatively less abundant (33 and 21 individuals respectively) and *Mustelus antarcticus* the least abundant of the species (18 individuals). It must be remembered however, that MaxN is an estimate of relative abundance, not absolute abundance, that is considered conservative (Cappo et al., 2003). Further, that our sampling was targeted towards mixed reef, which is patchily distributed and comprises approximately 30% of the FMP shelf area (Hill et al. 2014).

Most species had a patchy distribution across the FMP shelf and the distribution of each species was distinct (Figure 2). Substrate type, depth and latitude ranged from explaining a small proportion of the variation in MaxN (e.g. 11% for *P. richardsoni*; Table 2) to explaining a substantial proportion for *H. percoides* (42%; Table 2) and *P. bassensis* (63%; Table 1). *Helicolenus percoides* was more abundant on mixed reef in the southern end of the Marine Park (Figures 2, 3). *Latris lineata* was very patchily distributed (Figure 2) and only found on a few mixed reef sites in high abundances, but did not vary significantly with depth or latitude (which only explained an additional 8% of the variation in MaxN; Table 2, Figure 3). *Nemadactylus macropterus* was more widespread (Figure 2) and MaxN was greater on mixed reef, in shallower waters and in the southern end of the Marine Park (Figure 3). *Platycephalus bassensis* was more abundant on sand and at shallower depths depth (Figure 3), while *M. antarcticus* and *P. richardsoni* were only found in low abundances and were not significantly related to any of the predictor variables.

Whilst not all fish observed at or around MaxN can be measured because some fish occlude the view of other fish, in our case a large proportion of the observed fish contributed to length metrics. Most individuals of *M. antarcticus, P. bassensis* and *P. richardoni* observed at MaxN could be measured, whilst 77%, 64% and 70% of *H. percoides, L. lineata* and *N. macropertus* individuals could be measured respectively. The length-frequency distributions of fish measured at MaxN also varied between the species (Figure 3). There is no legal size limit for *H. percoides* and most individuals were between 200 and 300 mm (average size = 223 mm), with some very small individuals (~ 100 mm). The majority of *L. lineata* and *M. antarcticus* recorded were juveniles below the legal size of 550 mm and 750 mm respectively. The length frequency of *Nemodactylus macropertus* appeared to be bimodal with peaks at 200 and 300 mm, just below and above the legal size limit of 250 mm. The majority *P. bassensis* and *P. richardsoni* individuals were much larger than the legal size limit for flathead species (Figure 3).

Overall the length of most species was not related to the measured environmental factors (Figure 4). The exceptions were *H. percoides* where individuals were significantly smaller on sand substrate (however, there were few individuals recorded on sand) and *M. antarcticus* where the size of individuals decreased with depth (Figure 4).

3.2. Design-based status estimates

The design-based estimates provide a present-day baseline for the status of potential indicator species. After taking into account the proportion of sand and mixed reef habitat to produce estimates for the entire Marine Park, *N. macropertus* was the most abundant species with approximately five individuals expected to be observed at MaxN on average in any BRUVs drop (Table 3). *Platycephalus bassensis* was the next most abundant species with approximately two individuals expected on average, while *H. percoides* and *M. antarcticus* were the least abundant with less than one individual expected to be observed at MaxN in any BRUVs drop (Table 3). However, many species were more abundant in one or other of the strata supporting the distribution results presented above. *Nemodactylus macropertus* was most abundant in the mixed reef strata with an average of approximately nine individuals expected to be observed at MaxN in mixed reef drops, and was still the most abundant species overall. *Latris lineata* and *H. percoides* were also most abundant in the mixed reef strata, however *L. lineata* is expected to be more abundant than *H. percoides* (Table 3). *Platycephalus bassensis* and *P. richardsoni* were more abundant in the sand stratum with approximately three and one individual expected at MaxN per BRUVs drop respectively (Table 3). Variance in these estimates of abundance, as measured by the 95% confidence intervals, was generally largest for species with higher abundances such as *N. macropertus* and *L. lineata* (on mixed reef).

3.3. Power analysis

In all cases, revisiting sites and focussing sampling within one of the strata (the preferred substrate type or habitat) would require the least sampling effort and be the most efficient strategy (Figure 6). As the magnitude of the effect size increases, the number of sampling sites required to detect a difference decreases. A 100% increase in mean MaxN should be detectable with a feasible amount of sampling effort (nominally < 100 sites at each sampling event for BRUVs –focussed surveys) for most species under one of the sampling scenarios. Species for which a smaller increase of 50% in mean MaxN should be detectable include *M. antarcticus* (all substrata revisted and on sand)*, P. bassensis* and *P. richardsoni* (on sand) and for *N. macropertus* and large fish (> 250 mm) on mixed reef (Figure 6).

4. Discussion

Australia's new Marine Park network covers a vast area, including regional representation of shelf waters. Despite this, little is known of the habitats found within them, or the species they support. Here, in the Flinders Marine Park, an good example of a park where we have little prior knowledge, we have investigated a practical approach to quantifying the abundance and distribution of key demersal fish species as both an initial inventory, and a baseline for future monitoring programs. For six potential indicator species, by combining BRUVs-based surveys with a two-stage GRTS-based sampling design we were able to: (1) quantitatively describe their distribution and characteristics in the Marine Park; (2) establish quantitative and representative estimates of their average relative abundance throughout the Marine Park to serve as a present-day baseline for future monitoring; (3) conduct power analyses to estimate the magnitude of increases that we may be able to detect with feasible levels of sampling effort. Our approach is transferrable to other regions where monitoring programs must be designed based on limited spatial and biological data, assisting with decisions on *what* and *how* to monitor.

4.1. Distribution of potential indicator species in the Marine Park

Each of the potential indicator species had a distinct distribution across the FMP that for some species was well described by substrate type, depth and/or latitudinal position. Substrate type was the most influential of these variables, significantly affecting the observed abundances of four of the species at BRUV sites as well as resulting in different habitat-specific baseline estimates of their average abundance across the entire FMP shelf. In line with previously reported associations (summarised in Table 1 and reported in Williams and Bax, 2001 for the coast north of our survey region), *H. percoides, N. macropertus* and *L. lineata* are more abundant in reef habitats, while *P. bassenis* is associated with sand habitats. The importance of substrate type for understanding the distribution of conservation values and indicators within the FMP implies that obtaining comprehensive maps of the distribution of substrate type and therefore habitats is ultimately an important goal. Mapping using multibeam sonar will produce such maps, but requires intensive coverage in relatively shallow shelf environments and will take many years to achieve. In the meantime, a probabilistic two-phase sampling program that first determines the prevalence and distribution of habitats can suffice for monitoring (Lawrence et al., 2015).

Depth, generally has a strong effect on the distribution of many marine species and consequently assemblages (reviewed by Brown and Thatje (2014), Wilmliams and Bax, 2001). In our study however, depth only influenced the relative abundances of *N. macropertus* and *P. bassensis*. This may be because the depth range sampled within the FMP (40 m to 175 m) was within preferred niche of most of the species (summarised in Table 1). However, this pattern may also have been affected by the highly patchy distribution of some species, such as *L. lineata,* or low relative abundances observed at MaxN for other species, such as *P. richardsoni,* may make it difficult to distinguish patterns. Our results suggest that in this region on the mid to outer shelf, stratifying BRUVs deployments by habitat may be more beneficial than stratifying by depth for our species of interest. Finally, the variance explained by our environmental variables was small for some species such as *M. antarcticus, P. bassensis* and *L. lineata* indicating that other unmeasured factors play an important role in the distribution of these species. These factors may include distance from reefs (Schultz et al., 2012), the size and complexity of reefs (Moore et al., 2011) or factors influencing food availability such as the interplay between upwelling, productivity and seafloor currents (e.g. Schultz et al. (2012)). While understanding these drivers may be important from an ecological perspective, they are likely to be more difficult to quantify than habitat type and therefore less useful for planning monitoring programs.

4.2. Size patterns of potential indicator species

For studies monitoring the response of protected areas to altered fishing effort, one of the key metrics has been changes in the abundance of larger fish (Denny et al. 2004, Barrett et al. 2007, Edgar et al. 2014). This is because smaller fish are protected by minimum size limits in the absence of high-grading, or larger fish are removed through size-based gear selectivity. In the FMP we cannot reliably relate our patterns to fishing pressure because we have no data from before the establishment of the Marine Park. However, the observed cumulative size-frequency distributions of many species fit with expectations. For example, a substantial proportion of individuals of the four species, *N. macropertus, P. bassensis, P. richardsoni* and *H. percoides*, previously targeted by demersal trawl in modest amounts of the FMP shelf (1,750- 2,470 kg/yr between 1985 and 2007; AFMA unpublished data) were above legal size and/or size at 50% maturity. Conversely, a high proportion of the two species that can still be fished on the FMP shelf, *L. lineata* and *M. antarcticus,* were sub-legal or sub-adults. This suggests that some effects of the Marine Park may already be observable; but there are plausible alternative explanations. For example, the recruitment of *L. lineata* can be highly variable with periods of sustained poor recruitment (Tracey and Lyle, 2005). An excess of small individuals may be observed after a strong recruitment event and the cohort approaches size at maturity. Never-the-less, as we have shown BRUVs are capable of collecting size frequency data for the majority of fish observed at MaxN. The resulting size distributions may be a useful indicator for detecting shifts that may occur in response to protection from some fishing gears when compared to present-day baselines and control areas outside the Marine Park that are subject to fishing, and is worth investigating further.

The length of individuals was rarely related to the three environmental variables we recorded; depth, substrate type and latitude. We may have expected some relationships with depth because most of the potential indicator species undergo ontogenetic shifts where juveniles are found in shallow waters and move to deeper waters as they mature (Jordan 2001, Tracey and Lyle 2005) and trawl surveys to the north of our survey region also found a tendency for larger individuals in deeper shelf waters (Williams and Bax, 2001). However, in our study only *M. antarticus* exhibited significant depth-related size patterns and smaller individuals were found in deeper waters. This trend has been observed in *M. antarcticus* caught by commercial long-lines in Western Australia (Braccini 2016), but the reason is unclear. We may not have found depth-related size patterns for the remainder of the species because of the small number of fish for some species (e.g. *P. richardsoni*). Previous studies have also aggregated data from many trawls into pre-defined depth bins and compared frequency histograms, whereas we analyse length-data at the level of each BRUV deployment. Difference in depth-related size patterns observed between our study and previous studies may therefore be due to the different gear type used, which can affect the strength of the relationships observed (Williams and Bax, 2001), or different analysis methods.

4.3. Evaluation of potential indicator species

Many factors go into choosing an appropriate indicator of change in marine systems (Hayes et al., 2015). Here we primarily focus on what magnitude of change in the abundance of our subset of demersal fish are likely to be detected through a monitoring program with a realistic amount of sampling effort (<100 deployments). Our power analysis suggests that by targeting sampling within species' preferred habitat, we expect to be able to detect a 50% increase in the abundance of *N. macropertus, M. antarcticus*, and the two species of flathead with reasonable sampling effort. However, for the flathead species, this was only under scenarios where the same sites are revisited through time. For *H. percoides* and *L. lineata*, around a 100% increase in abundance would be required to detect change within practical sampling constraints. The changes that we expect to be able to detect are small to modest, compared to other studies where a four-fold increase in the abundance of some species (Barrett et al., 2007) and up to 17- fold increase in the abundance of legal size snapper in NZ marine reserves (Smith et al., 2014) has been observed after 10 and 14 years' protection respectively. This makes us confident that we would be able to detect biologically meaningful changes within the reserve, should they occur.

Overall, all species examined here are representative of the demersal fish communities within the Tasmanian Shelf and Southeast Shelf Transition biogeographic provinces, and protection of these is a key listed objective of the FMP (Commonwealth of Australia 2006, Director of National Parks 2013). Each of the species may provide insight into different aspects of the performance of the Flinders Marine Park. For example, *N. macropertus* and both flathead species are essentially protected from fishing in the AMP and may be suitable indicators for species expected to recover, while species such as *M. antarcticus* and *L. lineata,* which are not fully protected, may be indicators of zoning within the FMP if future management plans give added protection. By following these populations through time, and contrasting Marine Park observations with observations sampled from adjacent fished areas, it is likely that a realistic BRUV-based monitoring program may effectively evaluate the effect that the Marine Park is having on a range of target and ecologically relevant species.

An unsurprising finding from our power analysis is that less sampling effort was required to detect changes when the same sites are revisited through time and when sampling was restricted to a species' preferred habitat. In both cases this is because the variance in abundance estimates is minimised. For most species, the sampling required to detect a small effect of 50% increase in abundance, should it occur, was only feasible under these circumstances. However, the most efficient monitoring design for one species that repeatedly samples the same sites in only one habitat type through time may not be the optimal design overall, depending on the objectives of the program. One example is where there might be interest in monitoring a suite of indicator species that occur in different habitats (e.g. *N. macropertus* and *M. antarcticus*). Another is where there is also interest in improving information on the spatial distribution of assets within the Marine Park. In the latter case, it may be better to revisit a proportion of sites and gain information on new sites in a rotating panel design (Gitzen et al., 2012). The sampling sizes required to detect changes in a rotating panel design would lie somewhere between the two scenarios we tested here and would depend on the specification of the panels. Finally, once some temporal data is available, more sophisticated power analysis or simulations that can incorporate temporal and other sources of variation would be useful to inform the frequency of sampling as well as the likely timeframe for which to observe trends, if they exist (Perkins et al., 2017). The effects of season on the movements of demersal fish should also be considered (e.g. Smith et al. (2014)).

4.4. Evaluation of sampling methods

In our survey, we trialled a relatively new approach to probabilistic sampling and adapted it to suit the logistics of sampling with BRUVs across large areas. The decision to implement the GRTS spatially balanced sampling design was guided by several needs including: accommodating the multiple objectives of the broader research program; ensuring good spatial coverage to enable observations from individual sites to scaled up to give representative estimates of the status of species across the entire FMP shelf; and providing a foundation for future monitoring efforts in the FMP. Using the GRTS sampling design and various sampling gears in two surveys has enabled: the quantification of the extent of coverage of individual shelf habitats (Lawrence et al., 2015) and the benthic biota they contain (Lawrence et al., 2015; Monk et al., 2016); an inventory and description of the distribution of demersal fish communities (Hill et al., 2014); and an assessment of the distribution and present status of key fish species (this paper) within the FMP. The GRTS approach allowed us to disproportionately target different substrates *a priori* and still gain representative estimates for the reserve by adjusting the inclusion probability of sites, which facilitates the compromises often necessary in survey designs. The GRTS approach was also flexible with the ability to adjust sampling, such as adding sites in our case, including new sites in the field as situations necessitate, while maintaining spatial balance and representiveness across the region of interest. One downside of the GRTS approach and its inherent spatial balance is that sites are ordered and there can be considerable travelling distances between sites when used to survey large Marine Parks such as the FMP and others within Australia's Marine Park network. This means that there is a trade-off in efficiency between the time it takes to travel to sites versus the time taken to deploy equipment. In our case, implementing a two-stage GRTs design, where we sampled clusters of sites around a central site, proved a feasible strategy for estimating the status of demersal fish. This strategy could readily be adopted for other components of the ecosystem or other regions where deployment times are short relative to steaming time.

One of the features of the GRTS methodology is that it uses a local variance estimator (Stevens and Olsen, 2003). The local variance estimator assesses variance only from nearby sampling locations, not from all sampling locations, meaning that it compensates for the spatial pattern of the survey. This generally results in estimates that are less variable, and more reasonable, than if they were calculated using simple random sampling variance (Stevens and Olsen, 2003). Both of these are advantageous for monitoring natural resources. The local variance estimator uses at least four GRTS samples per strata (and in our case each stage 2 cluster) and so four samples per cluster are needed optimise variance estimates. This should be taken into account when designing sampling programs, especially in cases where there are likely to be non-responses due to equipment failure (as occurred with some of our BRUVs failing to record) or some other inability to sample.

A key reason why we did not exclusively use judgemental sampling for this study, as is commonly done in Marine Protected Area assessments, is that we wanted to ensure our estimates were representative of the entire Marine Park and to take into account the proportion of each habitat type in the Marine Park. After taking into account the availability of habitats, *Nemodactylus macropertus* was the most abundant potential indicator species across the FMP and *Helcolenus percoides* and *Mustelus antarcticus* were the least abundant.

We used the mean MaxN that we would expect to observe in any deployment across the FMP as our park-scale metric of abundance. We chose this instead of total MaxN because MaxN can be asymptotic (Stobart et al., 2015) and therefore total MaxN may underestimate relative abundance when scaled to the entire Marine Park. In addition, it is important to recognise that because MaxN measures relative abundance, values will be much less than those recorded via destructive sampling methods (e.g. trawls). Never-the-less BRUVs are becoming a standard tool for sampling demersal fish (e.g. Fitzpatrick et al. (2012); Langlois et al. (2012)) and we have demonstrated theirability to quantify baselines for fish that are attracted to baits, which is often the subset that we are interested in. As the range of metrics extracted from the video data increases (Stobart et al., 2015), there will be a need to evaluate each of these metrics for their relative effectiveness in detecting trends through time and differences between regions.

5. Conclusions

Here we have demonstrated an approach to develop sampling programs to inventory and monitor demersal fish over large areas where we have little existing data. We found that several target fishery species were encountered in sufficient numbers to form a core indicator group by which to evaluate the effectiveness of a Marine Park. Power analysis suggested that for most of these, detecting increases in abundance as small as 50% from present values should be feasible if sampling is restricted to a species' referred habitat and the same sites are sampled through time. Additionally, the abundance of large fish may also be a suitable indicator. Adopting a based spatially balanced sampling-design had several advantages. It was flexible in maintaining spatial balance in a range of field scenarios and the two-stage implementation minimised transit times. Because GRTS is a probability sampling design, we were also able to scale up individual BRUV abundance estimates for each of our species to the entire Marine Park. These estimates take into account the proportion of available substrate and give us greater confidence in the generality of patterns detected. When coupled with sampling programs that contrast temporal abundance patterns in Marine Parks with adjacent fished locations this approach should allow us to evaluate the response of such Marine Parks with varying levels of protection from fishing activities. Although we have focussed on one particular park in Australia's Marine Park network, our approach that combines spatially-balanced probabilistic sampling with observations from BRUVs, should be applicable for aiding decisions on what and how to monitor in any demersal region where habitat and biological data are limited.

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University, University of Western Australia, NSW Office of Environment and Heritage, NSW Department of Primary Industries and the Integrated Marine Observing System.

Tables:

	Distribution and Ecology						Fisheries			
Species name (Common name)	**Distribution**	**Habitat**	**Depth range**	**Lifespan**	**Trophic ecology**	**Ontogenic habitat use**	**Fishery**	**Gear Type**	**Target/ Incidental**	**Gear Type allowed in CMR IUCN VI?**
Helicolenus percoides (Reef ocean perch)	Southern Australia and New Zealand	Reef and sand	50 -750 m	up to 42 years	Carnivores	No	Commercial	Demersal trawl	Targeted	No
								Danish seine	Targeted	No
Latris lineata (Striped trumpeter)	Southern hemisphere temperate waters	Reef	5 -400 m	up to 40 years	Higher Carnivores	Juveniles in shallow waters	Commercial	Hook and line Gillnet	Target	Yes
							Recreational	Hook and line	Targeted	Yes
Mustelus antarcticus (Gummy shark)	Endemic to Southern Australia	Not described	20-150 m	up to 16 years	Higher Carnivores	Juveniles aggregate	Commercial	Demersal gillnet	Targeted	Yes- above 183 m
								Longline	Targeted	Auto- No
								Demersal trawl	Incidental	No
							Recreational	Hook and line	Targeted	Yes
								Gillnet	Targeted	Yes- above 183 m
Nemodactylus macropertus (Jackass morwong)	Southern hemisphere temperate waters	Reef and reef edge	20 -450 m	up to 16 years	Carnivores	Juveniles inshore	Commercial	Demersal trawl	Targeted	No
								Danish seine	Incidental	No
							Recreational	Hook and line	Incidental	Yes
Platycephalus richarsoni (Tiger flathead)/ *Platycephalus bassensis* (Sand flathead)	Eastern and South-eastern Australia/ Southern Australia	Un-consolidated sediments	10- 400 m/ 5- 100 m	up to 15 years	Carnivores	Juveniles inshore	Commercial	Demersal trawl	Targeted	No
								Danish seine	Targeted	No
								Demersal gillnet	Incidental	Yes- above 183 m
							Recreational	Hook and line	Targeted	Yes

Table 1. Distribution, ecology and fisheries information for potential indicator species.

Information sourced from: Kailola et al. (1993), Tracey and Lyle (2005); Tracey et al. (2007), Jordan (2001), Froese and Pauly (2017) , AFMA (2017).

Species	Deviance Explained	Predicted Zeros	Observed Zeros
Helicolenus percoides	0.42	0.65	0.69
Latris lineata	0.08	0.77	0.78
Mustelus antarcticus	0.12	0.67	0.67
Nemadactylus macropterus	0.29	0.36	0.41
Platycephalus bassensis	0.63	0.75	0.76
Platycephalus richardsoni	0.11	0.73	0.75

Table 2. Goodness of fit statistics for models relating MaxN at each BRUV site to substrate type, depth and latitude.

Goodness of fit statistics are estimated from negative binomial models for all species except *M. antarcticus* which used a binomial model. The proportion of deviance explained and the proportion of zeros predicted are the average from 10,000 Bayesian bootstraps of the data. The proportion of observed zeros is calculated from the BRUVs data.

Species	Strata	n	Mean	SE	L95	U95
Helicolenus percoides	All	42	0.51	0.20	0.12	0.90
	Mixed	30	1.39	0.56	0.28	2.50
	Sand	12	0.10	0.12	0	0.33
Latris lineata	All	42	0.75	NA	NA	NA
	Mixed	30	2.33	1.04	0.28	4.38
	Sand	12	0.00	NA	NA	NA
Mustelus antarcticus	All	42	0.51	0.14	0.23	0.79
	Mixed	30	0.21	0.08	0.06	0.36
	Sand	12	0.65	0.21	0.24	1.06
Nemadactylus macropterus	All	42	4.83	1.81	1.28	8.38
	Mixed	30	9.26	2.01	5.32	13.20
	Sand	12	2.74	2.49	0	7.62
Platycephalus bassensis	All	42	1.94	0.73	0.51	3.37
	Mixed	30	0.09	0.06	0	0.21
	Sand	12	2.82	1.07	0.71	4.92
Platycephalus richardsoni	All	42	0.73	0.26	0.23	1.23
	Mixed	30	0.36	0.17	0.03	0.69
	Sand	12	0.90	0.37	0.18	1.63

Table 3. Design-based estimates (and uncertainty) of the average MaxN for each species across all strata and for each strata individually across the entire Flinders CMR shelf.

Estimates were generated from the two-stage sampling design using BRUV drops at GRTS sites. Estimates represent the average (Mean), standard error (SE) and 95% confidence intervals (L95, U95) number of individuals expected to be observed at MaxN in any one drop on anywhere on the FMP shelf (all strata) on mixed reef substrate on the FMP shelf (Mixed strata) and on sand substrate (Sand strata) on the FMP shelf.

Figures:

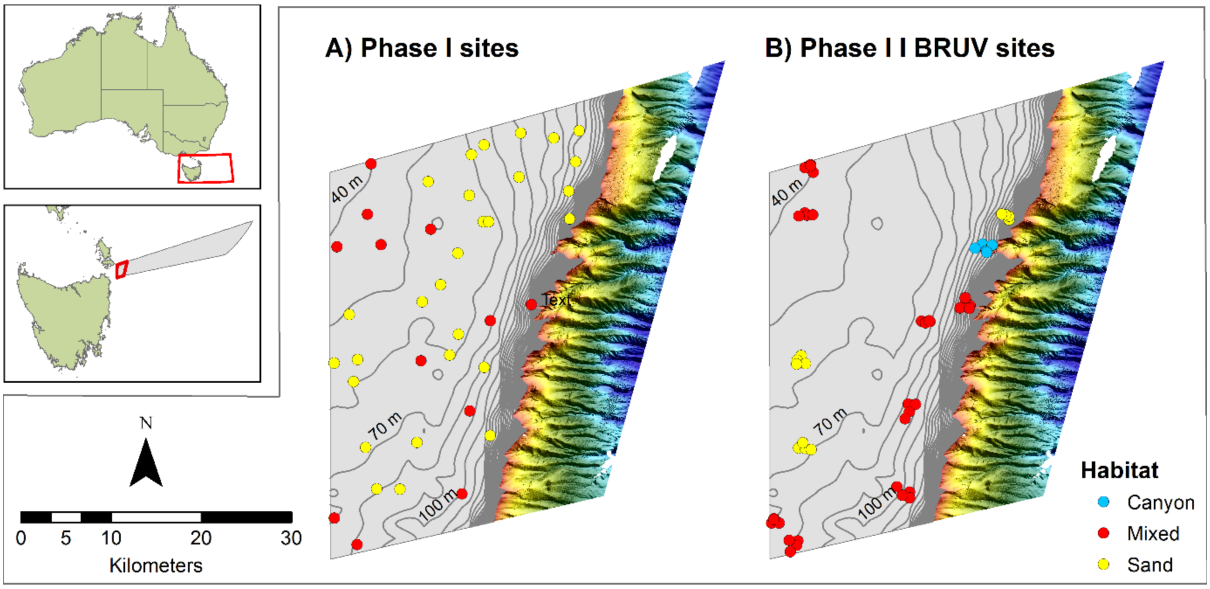

Figure 1. Location of the Flinders CMR, off north-east Tasmania. A) Phase 1 GRTS site sampled and classified as mixed reef or sand substrate. B) Phase 2 BRUV sampling which involved a 2-stage design where BRUV sites were clustered around a subset of the Phase I sites. Sites are colour-coded according to the substrate type in Phase 1. The blue sites indicate a cluster of sites surrounding the preferentially chosen canyon head site.

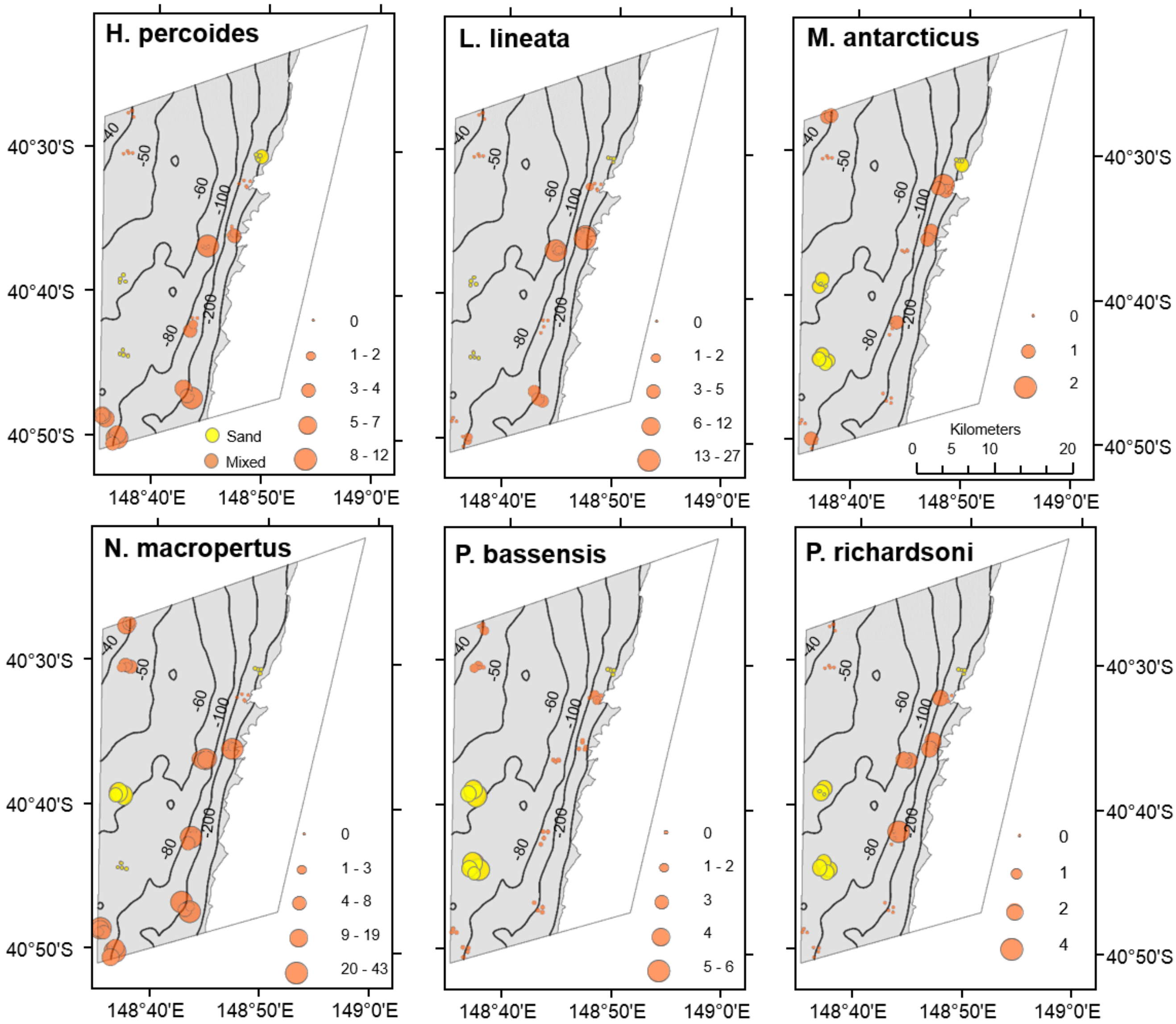

Figure 2. Distribution and abundance of potential fish indicators on the Flinders CMR shelf based on observations from Baited Underwater Video (BRUV) deployments. Yellow symbols indicate deployment on sand, while orange symbols indicate deployment on mixed reef.

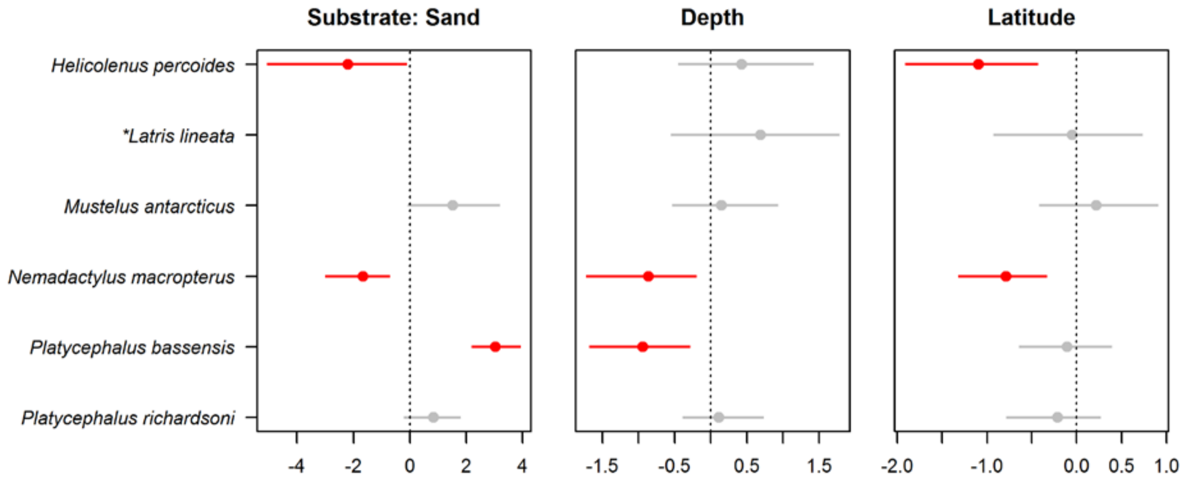

Figure 3. Mean coefficient values and 95% confidence intervals for the effect of environmental variables on the abundance of potential indicators species.

Coefficients and Confidence Intervals (CIs) were determined by 10,000 Bayesian bootstraps of the data and a negative binomial model. Red bars indicate variables whose 95% CI do not overlap with zero and are influential for each species. *Substrate was not included in models for *Latris lineata* because it only occurred on mixed reef substrate.

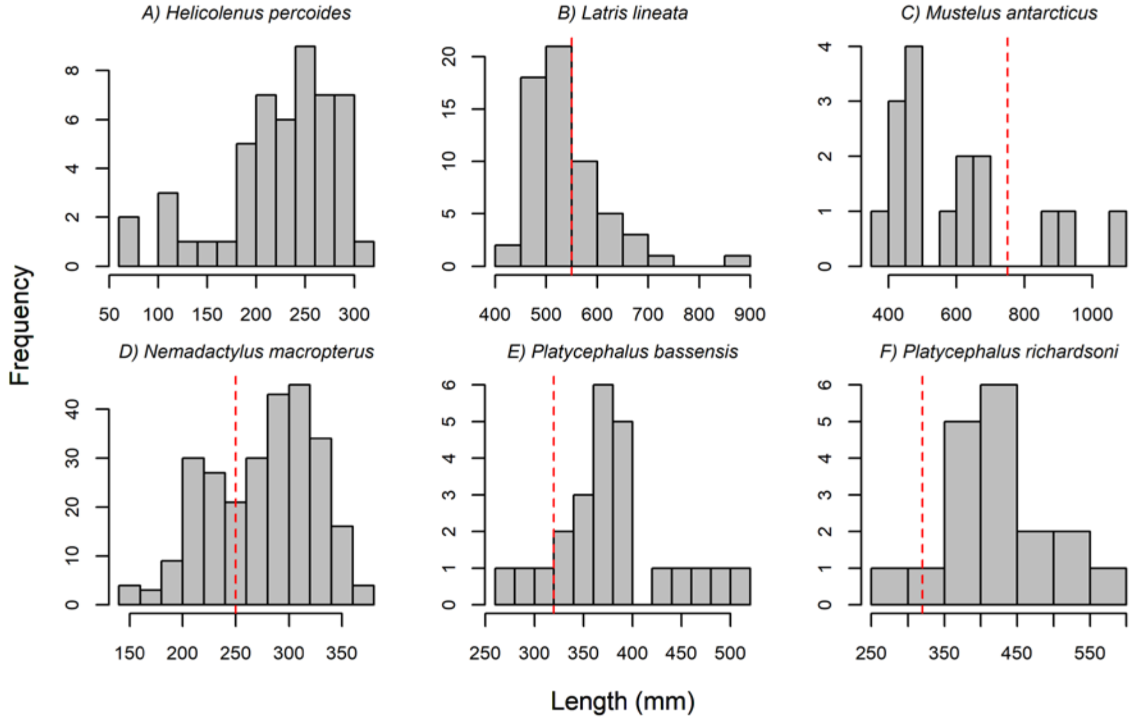

Figure 4. Length frequency of fish measured at MaxN.

Red dotted lines indicate the Tasmanian Recreational Fishery legal size for each species.

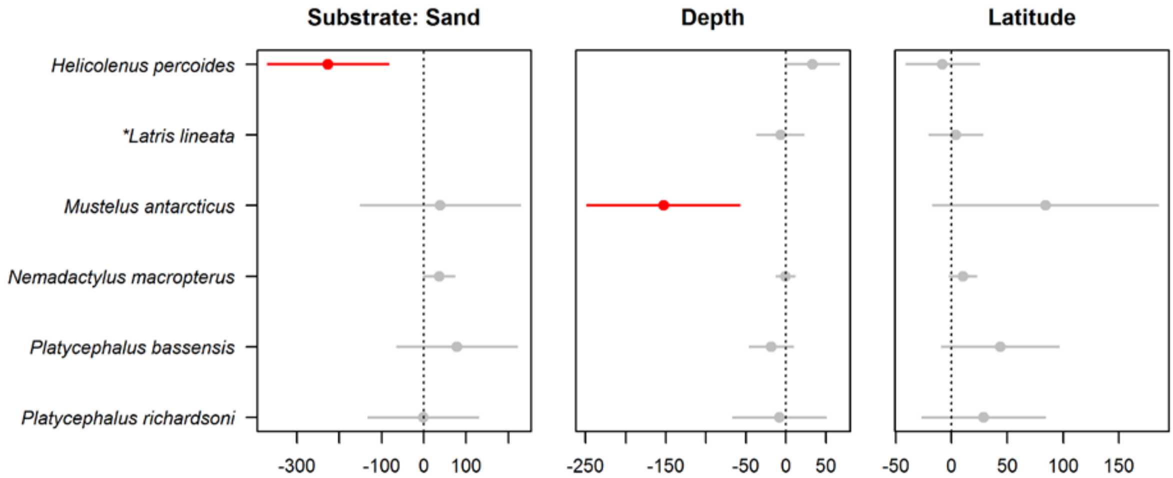

Figure 5. Mean coefficient values and 95% confidence intervals for the effect of environmental variables on the length of potential indicators species.

Coefficients and Confidence Intervals (CIs) were determined by 10,000 parametric bootstraps of the data and a linear model (*M. antacrticus, P. bassensis, P. richardsoni*) or linear mixed effects model. Red bars indicate variables whose 95% CI do not overlap with zero and are influential for each species. *Substrate was not included in models for *Latris lineata* because this species only occurred on mixed reef substrate.

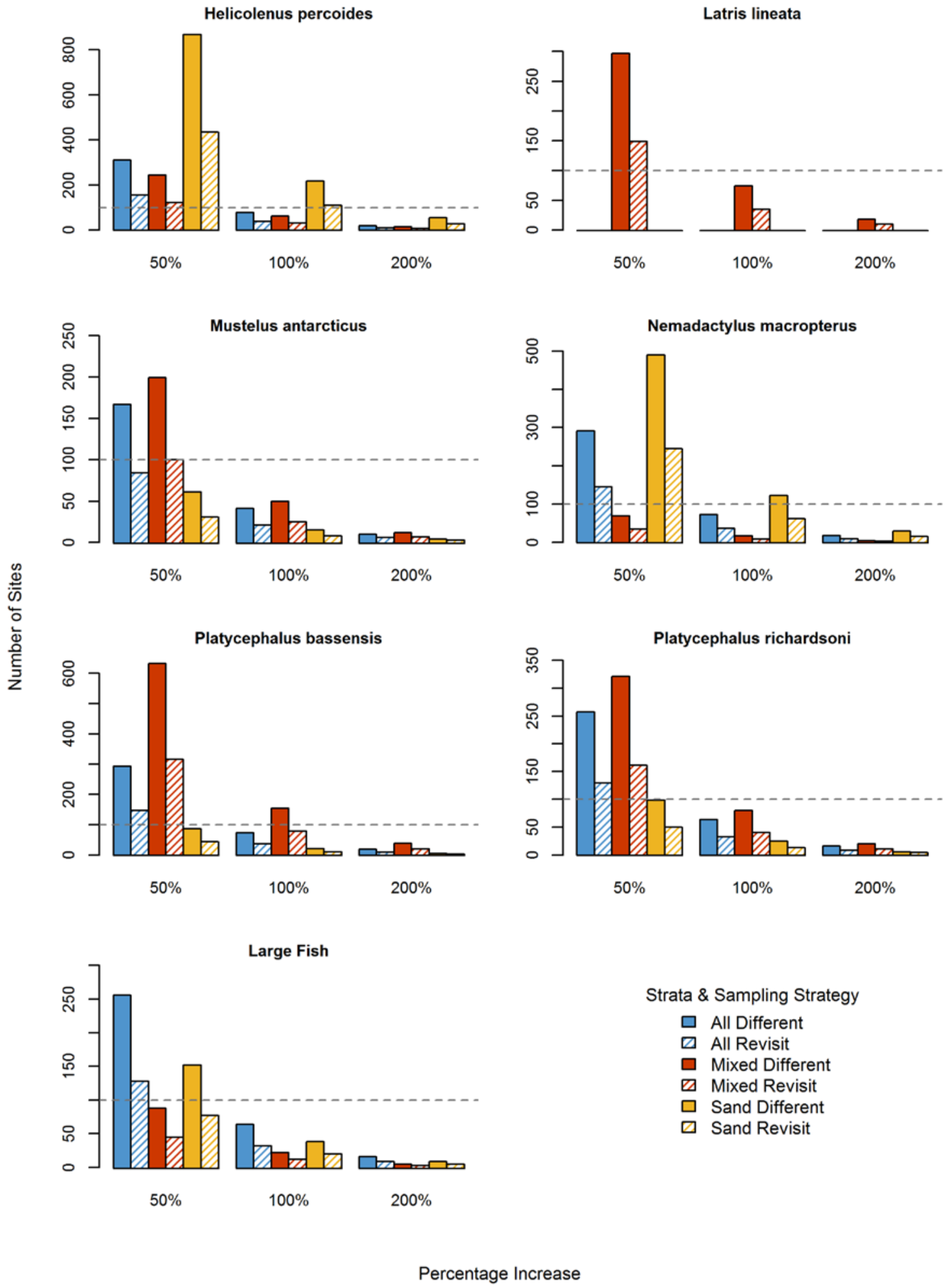

Figure 6. Power analysis to estimate the approximate number of BRUV sites required to detect 50, 100 and 200% increase in mean MaxN across the Flinders CMR between two sampling events.

Sample size (number of BRUV sites) are estimated separately for sampling: all strata (blue); only mixed strata (red) and only sand strata (yellow) and for two sampling strategies: the same sites are revisited in the second sampling event (hashed fill) or new and different sites are revisited (solid fill). The dashed grey line indicates the 100 samples considered a 'feasible' amount of sampling effort.

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