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Key drivers of effectiveness in small marine protected areas

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Abstract Marine Protected Areas (MPAs) are a key management tool for the conservation of biodiversity and restoration of marine communities. While large, well-designed and enforced MPAs have been found to be effective, results from small MPAs vary. The Hawkesbury Shelf, a coastal bioregion in New South Wales, Australia, has ten small, nearshore MPAs known as Aquatic Reserves with a variety of protection levels from full no-take to partial protection. This study assessed the effectiveness of these MPAs and analysed how MPA age, size, protection level, wave exposure, habitat complexity, and large canopy-forming algal cover affected fish, invertebrate and benthic communities. We found aspect, protection level, complexity and algal canopy to be important predictors of communities in these MPAs. Most MPAs, however, were not effective in meeting their goals. Only full no-take protection (three out of ten MPAs) had a significant impact on fish assemblages. One no-take MPA—Cabbage Tree Bay—which is naturally sheltered from wave action and benefits from an active local community providing informal enforcement, accounted for

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most of the increased richness of large fish and increased biomass of targeted fish species. Our findings suggest that small MPAs can enhance biodiversity and biomass on a local scale but only if they have full no-take protection, are in sheltered locations with complex habitat, and have positive community involvement to engender support and stewardship. These results provide a baseline for robust assessment of the effectiveness of small MPAs and inform future management decisions and small MPA design in other locations.

Introduction

Anthropogenic impacts on marine and coastal ecosystems, such as pollution, construction and erosion, climate change, biological invasions and over-exploitation of natural resources (McKinley et al. 2011; Newton et al. 2012; Mayer-Pinto et al. 2015) are increasing with wide-ranging consequences for biodiversity and ecosystem functioning (Doney et al. 2011). In the face of multiple stressors, Marine Protected Areas (MPAs) are recognised as a management tool to conserve and facilitate recovery of marine ecosystems, and many studies have addressed their design and benefits (Claudet et al. 2008; Edgar and Stuart-Smith 2009; Russ et al. 2015). While large, well-designed MPAs and networks of MPAs have been found to be effective in increasing species richness and biomass (Edgar et al. 2014), results from small MPAs are varied (Bonaldo et al. 2017). Yet globally, most MPAs are small and vary in their level of protection; 60% are 1 km² or less; and 94% of these allow fishing (Costello and Ballantine 2015).

Effective MPA design involves consideration of location characteristics, coverage of representative habitats, size, spacing and distribution necessary to sustain basic life-cycle functions (Dugan and Davis 1993; Fox et al. 2012), and is in theory best implemented as either a large MPA or a network of integrated smaller MPAs (Friedlander et al. 2003). Despite some MPAs having been designed using systematic spatial conservation planning approaches, most are not designed in a systematic way (Agardy et al. 2011; Martín-García et al. 2015), and there are concerns that the MPA label is creating an illusion of conservation of marine biodiversity (Lynch 2006; Costello and Ballantine 2015). Ineffective MPAs are thought to be under-reported in the literature, leading to lack of understanding of causative factors such as poor design or enforcement (Guidetti et al. 2008).

In Australia, 85% of people live within 50 km of the coast (ABS 2004). The marine estate in New South Wales (NSW), Australia, which includes ocean, estuaries, wetlands, coastline, lakes and islands (MEMA 2016), is of high social and economic value. It bestows a range of benefits including recreation, enjoyment, cultural heritage, employment and intrinsic value (Jordan et al. 2016). The Hawkesbury Shelf is one of the five marine bioregions wholly in NSW but the only one without a large multi-zone MPA. It is also where NSW's largest city, Sydney, and arguably the world's most biologically diverse natural harbour—Sydney Harbour—are located (Johnston et al. 2015). It encompasses estuarine, coastal and offshore waters, and is home to ~ 7.5 million people—equivalent to almost a third of the population of Australia (ABS 2016). The bioregion's environmental assets include biologically-diverse ecosystems composed of a range of habitats, assemblages, rare and protected species (Breen et al. 2005, Johnston et al. 2015; MEMA 2016). Coastal subtidal rocky reefs in the bioregion, which are the focus of this study, comprise large

canopy forming macroalgae, urchin barren and sponge garden habitats, although the distribution and abundance of subtidal habitats has not been consistently documented (MEMA 2016).

Currently the Hawkesbury Shelf bioregion contains ten small coastal MPAs (locally known as Aquatic Reserves) all of which are 3 km^2 or less, and most of which are 1 km^2 or less. These MPAs were put in place to protect habitat and biodiversity and allow recovery of degraded fish stocks (Smith and Pollard 1996), yet they have different policies ranging from full no-take MPA (only three MPAs) to the protection of some invertebrates only—allowing harvesting of all fish and targeted invertebrates such as abalone and lobsters (NSW Government 2015; DPI 2016a). The Sydney region has a high number of recreational fishers with relatively high fishing effort and a high proportion of undersize fish being caught (Ghosn et al. 2010; West et al. 2015), so protection from over-fishing is important. Apart from spatial protection, important existing management strategies include size and bag limits and restrictions on fishing methods, together with public education and enforcement (DPI 2016b).

Low level or partial spatial protection, as found in most of the region's MPAs, has generally been found to be ineffective in conserving biodiversity and biomass (Edgar et al. 2014; Costello and Ballantine 2015); although recent studies in other regions have found some positive effects (Giakoumi et al. 2017). Furthermore, such complex and varied regulations can potentially be confusing for the greater public using these marine resources, leading to problems with enforcement and compliance and further compromising the desired protection (Ballantine 2014). Despite their anecdotal ecological and social value, we are aware of no comprehensive study of the effect of protection measures on fish, invertebrate and benthic assemblages in the Hawkesbury Shelf bioregion Aquatic Reserves.

There are numerous factors that may influence community assemblages of rocky reefs, and global studies have confirmed five key factors important to the effectiveness of MPAs, indicated by the acronym NEOLI: No-take, Enforced, Old (10 years or more), Large (100 km^2 or more) and Isolated by sand or deep water (Edgar et al. 2014). Such studies may be of limited application to small MPAs, however, where MPA size may not enclose the full range of habitat needs of a species (Bonaldo et al. 2017) and isolation cannot be achieved due to shallow reef extending across the MPA boundary. Other factors such as wave exposure and habitat quality—structural complexity, both biotic and abiotic—can also impact on fish and invertebrate communities (Friedlander et al. 2003; Beck et al. 2016; Malcolm et al. 2016). Small MPAs are popular as they are relatively easy to implement, yet there is limited knowledge of the effectiveness of small MPAs particularly on temperate rocky reefs. Positive outcomes have been reported, for example for snapper (*Chrysophrys auratus*) in the no-take Leigh Marine Reserve in New Zealand (Willis et al. 2003) and populations of commercial fishes in the Mediterranean and Canary Islands (Claudet et al. 2008), but in other studies results have been mixed (Edgar and Barrett 2012). Small temperate MPAs need to be better understood to be effective in meeting management and conservation goals.

This study evaluates the effectiveness—richness, abundance and biomass—of small MPAs as a function of their level of protection, size, age, habitat structural complexity, wave exposure (represented as aspect) and cover of large canopy-forming algae. We use a comprehensive data set encompassing twenty sites in the Hawkesbury Shelf bioregion, sampled at the same time of year over a three year period, under relatively consistent environmental conditions while allowing replication over space and time. We expect that no-take MPAs would have higher richness, abundance and biomass of targeted species than sites with a lower level of protection from fishing, and that sites with low wave-exposure

would have higher fish species richness compared to sites with higher wave exposure (e.g. Lester et al. 2009; Depczynski and Bellwood 2005; Beck et al. 2016). We discuss the results in the context of informing future management decisions, and contributing to the understanding of the drivers of effectiveness of small MPAs.

Method

Study area and sites

The Hawkesbury Shelf bioregion extends across almost 2° of latitude along the South Eastern Australian coast (32°54′S–34°34′S). We studied eight of the ten Aquatic Reserves located in the Hawkesbury Shelf bioregion; two Aquatic Reserves were excluded as they are estuarine rather than coastal. In addition, we included one National Park Marine Extension in the bioregion with a no-take policy, Bouddi, and one no-take MPA just south of the
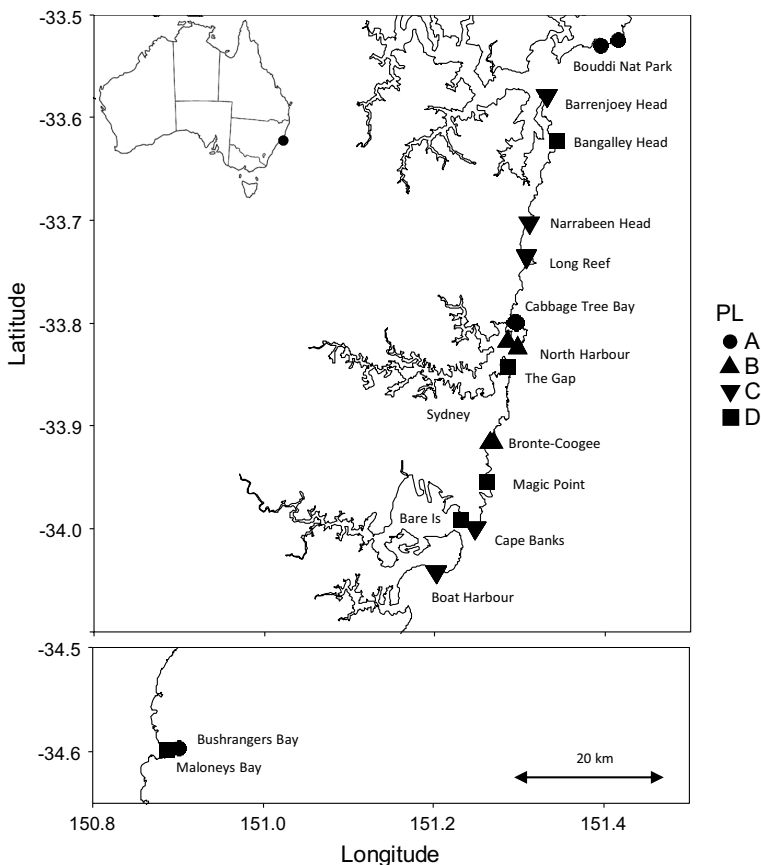
Fig. 1 Study locations by Protection Level (PL): A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA

Table 1 Site characteristics of MPAs of the Hawkesbury shelf bioregion

PL	Site	MPA	Age (years)	Size (km²)	Habitat complexity	Aspect	Algal canopy (%)	No. of surveys
A	Caves Bay	Bouddi National Park	46	2.82	0.76	0.90	8	4
	Bass Point	Bushranger's Bay	35	0.3	0.48	0.33	13	4
	Fairy Bower	Cabbage Tree Bay	15	0.2	0.70	0.10	23	6
	Maitland Bay	Bouddi National Park	46	2.82	0.42	1.00	48	4
	Shelly Beach	Cabbage Tree Bay	15	0.2	0.40	0.10	19	6
	Shelly North	Cabbage Tree Bay	15	0.2	0.58	0.10	0	6
B	South Clovelly	Bronte-Coogee	15	0.4	0.66	0.78	47	2
	Gordons Bay	Bronte-Coogee	15	0.4	0.44	0.40	35	5
	Inside North Head	North Harbour	35	2.6	0.54	0.33	11	4
	The Blocks	North Harbour	35	2.6	0.84	0.78	0	6
C	Boat Harbour	Boat Harbour	15	0.7	0.46	0.72	6	4
	Cruwee Cove	Cape Banks	15	0.2	0.34	0.44	19	4
	Dragon Alley	Barrenjoey Head	15	0.3	0.62	0.39	21	4
	Long Reef Basin	Long Reef	37	0.8	0.38	0.50	17	4
	Narrabeen Head	Narrabeen Head	15	0.1	0.24	0.56	60	1
D	Bangalley Head	n/a	n/a	n/a	n/a	0.80	0.50	2	2
	Bare Island	n/a	n/a	n/a	0.50	0.10	1	4
	Maloneys Bay	n/a	n/a	n/a	0.30	1.00	25	4
	Magic Point	n/a	n/a	n/a	0.90	0.39	35	2
	The Gap	n/a	n/a	n/a	0.64	0.72	55	4

Habitat complexity is measured on a scale from 0 (flat rocky reef) to 1 (complex cliffs and overhangs). Aspect is a proxy for wave exposure, measured on a scale from 0 (sheltered bay) to 1 (fully exposed shore). Protection Levels (PL): A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA

bioregion, Bushrangers Bay (Fig. 1), which in practice were equivalent to no-take Aquatic Reserves.

A total of 20 sites, distributed among 15 locations, each with one of four levels of protection, were surveyed (Fig. 1, Table 1). A total area of 40,000 m² was surveyed over a three-year period. Each MPA was designated as a single location containing between one and three sites, depending on MPA size and accessibility. Non-MPA sites were chosen to match nearby MPA characteristics such as aspect, benthic complexity and habitat as much as possible.

Habitat complexity of the surveyed sites varied from predominantly flat rocky reef to steep walls with caves and overhangs. Algal canopy ranged from urchin barrens (no canopy) to widespread kelp beds. Aspect, a proxy which we used to indicate exposure or wave

energy, varied from sheltered coastal bays and areas behind an island to fully exposed shorelines. Sites were separated by at least 300 m or a habitat gap such as an expanse of bare sand in accordance with Reef Life Survey guidelines (RLS 2016).

Data collection

During a 4-month period in the late Austral summer and autumn of 2015–2017, a total of 80 Underwater Visual Census (UVC) surveys were conducted by scientists and volunteers on SCUBA using the RLS methodology (RLS 2016). One of the three sampling periods fell within an El Niño, but the strength of effect was rated weak to moderate (BoM 2017). Survey sites were selected to encompass the range of our predictor variables as far as possible, considering access, topographic and other practical considerations. Whilst researcher surveys could be directed to manage sampling effort, volunteers were free to survey sites as often as they chose. Thirty-eight surveys were completed in 2015, nine in 2016 (when adverse conditions limited our surveys) and 33 in 2017. The RLS program is based on highly-trained volunteers and has been previously used to assess effectiveness of MPAs around the world (Edgar and Stuart-Smith 2014). Studies of this protocol showed no significant difference in the results produced by trained scientific divers compared to RLS volunteers (Edgar et al. 2004).

For each survey site in a given year, one to two 50 m transects were laid along consistent depth contours, depending on conditions, site specifics and gradients (Table 1). In order to exclude depth effects from our study, multiple depths within a narrow range (3–11 m) were used, in keeping with similar studies (Stuart-Smith et al. 2015). On average, three different depths were surveyed per site, achieving an overall average depth of 6 m.

Marine animals were recorded by divers swimming along paired blocks on either side of each transect tape. All fish species were counted and sized within a 5 m wide×5 m high window in front of the diver ("RLS method 1"). Mobile macroinvertebrates were then recorded within 1 m of each side of the transect tape ("RLS method 2"). Biota which could not be identified in situ were photographed and identification was subsequently confirmed by marine scientists or taxonomists. Twenty photo-quadrats of the substrate were also recorded; quadrats were placed according to a stratified random pattern along each transect (one photo every ~2.5 m) to quantify benthic biota. Complexity was evaluated using a visual estimation method whereby sites were placed on a continuum from 0 to 1, where 0 = no complexity and 1 = steep walls of 3 m or more, overhangs and caves. The relative position of sites on the continuum was refined over multiple surveys and was guided by the development of semantic indicators on the scale, for example 0.6 = large boulders and vertical drops between 1 and 3 m.. This approach allowed the complexity of each survey site to be evaluated and compared.

Data processing and analysis

Fish and invertebrate species richness and abundance were derived directly from method 1 (fish) and 2 (invertebrate) data respectively. Large fish were identified as fish with a recorded length of 20 cm or greater (Stuart-Smith et al. 2017). Fish biomass was calculated from size-classed abundance data using constants from the allometric growth equation Biomass = aL^b (Froese and Pauly 2017). Targeted fish were identified using the NSW

Recreational Saltwater Fishing Guide (DPI 2016b). Three small, non-targeted, widespread and abundant fish species mado (*Atypichthys strigatus*), eastern hulafish (*Trachinops taeniatus*) and eastern pomfret (*Schuettea scalaripinnis*) were excluded from the "all fish" abundance data prior to analysis to avoid masking effects of less abundant species on fish communities. For consistency, these species were also removed from the all fish abundance univariate analysis. Their removal did not alter the significance of protection level or the best model, but did result in the loss of significance for aspect and complexity. The results of the analyses including all species are presented in Supplementary material (Supplementary Tables S1, S2).

Benthic (photo quadrat) images were uploaded to CoralNet (Beijbom et al. 2012), and annotated using the Collaborative and Annotation Tools for Analysis of Marine Imagery (CATAMI) guide version 1.2 to the morphotaxa level (Althaus et al. 2013). We classified to this level as we were interested in the functional role of large canopy-forming macroalgae and coarse-level taxonomy (Althaus et al. 2015). Twenty images per transect were annotated with 25 random points per image for the calculation of benthic biota relative abundance and richness (Ayroza et al. 2015). Algal canopy was calculated from the relative proportion of annotation points in the photo quadrats for the two local canopy-forming species, kelp (*Ecklonia radiata*) and crayweed (*Phyllospora comosa*).

Protection level was represented as a categorical variable and complexity as a continuous variable, as described above. Aspect was used in this study as an approximation of wave exposure, as the prevailing large swell and storm direction is from the south–southeast (Short and Trenaman 1992; Kulmar 1995; Fulton and Bellwood 2004). Wave conditions were assumed to be consistent across the bioregion based on consistent extreme wave heights (varying from eight to nine metres across NSW) and direction (south-east to south–south-east in central to southern NSW) (Shand et al. 2011). Wave exposure was represented as a continuous variable from 0 to 1, indicating the direction faced by the shoreline nearest to the site, where 1 represents a shoreline facing directly SSE, i.e. maximum exposure. Sites inside small sheltered bays have shoreline on multiple sides, so this variable was assigned a value of half the aspect of the mouth of the bay, in order to account for the reduced wave exposure inside the bay compared to the mouth.

Statistical analyses

To investigate the relationship between predictor variables (age, size, protection level, aspect, complexity and algal canopy) and response variables (richness, abundance and biomass) we used linear mixed models in R (R Core Team 2015) as implemented in the *lme4* package (Bates et al. 2013). Random effects for year and site within location were included to account for non-independence across years, unbalanced sampling effort and possible spatial autocorrelation between nearby sites (Pinheiro and Bates 2000). Correlations between predictor variables, distributions of residuals, normality and over-dispersion were checked using the R cor() function, residual plots and QQ plots, and spatial autocorrelation was checked using correlograms in the *ncf* package (Bjørnstad 2004), and were all found to be acceptable in the final mixed model.

The small number and size of MPAs in the bioregion presented a challenge for the survey design, particularly with regard to achieving sufficient spatial replicates for the full range of predictor variables and the potential for interactions between them. We chose advanced modelling methods and analyses to account for the shortage of replicate MPAs (particularly A, no-take), limited geographic options (particularly aspect) and unbalanced sampling effort (determined by volunteer site preferences).

Twenty-three initial models were constructed to represent a range of ecologically-plausible predictor variable combinations including two-way interaction terms. Size and age only apply to MPA sites, so these were modelled on 15 sites whereas protection level, aspect, complexity and algal canopy were modelled across all 20 sites. Gaussian, Poisson and Negative Binomial distributions, and square root and log transforms of abundance and biomass were evaluated. Models were selected based on residual plots (over- and under-dispersion), normality and factor significance. MPA size and MPA age, and interactions complexity x protection level and algal canopy x protection level did not improve model fit and were not significant (p > 0.05) for any response variables. The final model used a Gaussian distribution on natural log transformed abundance and biomass data and untransformed richness data, with predictive variables for protection level, aspect, their interaction, complexity, algal canopy and random effects for year, and site within location.

The Information Theoretic approach, implemented in the *MuMIn package* in R (Bartoń 2013) was then applied to the global model developed in this initial analysis (Burnham and Anderson 1998). Models were ranked using corrected AIC (AICc) to allow for the relatively small number of sites (Symonds and Moussalli 2011). Predictor variables were assessed using Relative Variable Importance (RVI), calculated as the sum of the Akaike weights of all the models which contain each variable, and significance, using model-averaged coefficients (Table S2). Due to the presence of the interaction term, post hoc analyses were conducted by re-levelling the protection categorical variable, enabling all protection levels to be compared to each other using corrected p values (Rice 1989). Results were plotted using *ggplot2* (Wickham 2011) and are presented in tabular form in Table 2.

Multivariate analyses of community compositions were conducted using R packages *vegan* (Oksanen et al. 2013) and *MASS* (Venables and Ripley 2002). Similarity Percentage (SIMPER) analyses identified which species were driving differences between different protection levels (Clarke and Warwick 1994) and non-metric multi-dimensional scaling (nMDS) plots were used to visualize the differences in community composition, and the species and factors driving community differences based on the results of the univariate analysis.

Results

Fish

Aspect and protection level were the most important predictors of fish species richness. Naturally sheltered (low aspect, low wave exposure), high protection level (no-take) sites had higher richness, but also higher variance (Fig. 2). For large fish, protection level was significantly different between levels A (no-take MPA) and C (MPA open to line and spear fishing).

Complexity was the most important predictor for all fish and large fish abundance, but was only significant for large fish. More complex sites had higher fish abundance (Fig. 3). Aspect was the most important predictor of targeted fish abundance; naturally sheltered sites had more targeted fish. Protection level was important but not significant for large and targeted fish abundance.

Complexity was the most important predictor and the only significant factor for both all fish and large fish biomass (Fig. 4). Fish biomass increased with complexity. The interaction between aspect and protection level was significant and the most important predictor

Table 2 Relative Variable Importance (RVI), calculated as the sum of the Akaike weights of all the models which contain this predictor, from 0 to 1

	Aspect	Protection level (PL)	Aspect×PL	Habitat complexity	Algal canopy
Richness					
All fish	1*	1	1	0.91	0.74
Large fish	1*	1* A–C	1	0.86	0.63
Target fish	0.99*	0.99	0.95	0.53	0.41
Invertebrates	0.96	0.96	0.92	0.60	0.45
Benthos	0.97	0.98	0.94	0.90	
Abundance					
All fish	0.83	0.25	0.18	0.92	0.50
Large fish	0.71	0.83	0.33	0.97*	0.52
Target fish	0.97*	0.91	0.80	0.41	0.33
Invertebrates	0.62	0.47	0.37	0.65	0.98*
Biomass					
All fish	0.88	0.17	0.07	1*	0.28
Large fish	0.88	0.75	0.47	0.96*	0.28
Target fish	0.99*	0.97* A–B, A–C	0.94* A–B	0.49	0.29

An RVI value of 1 indicates this variable is present in every top fitting model

*Predictor is significant in the model averaged coefficients, $p < 0.05$. Letters indicate categorical variable pairs which are significantly different e.g. A–C = no-take MPAs are significantly different to MPAs allowing line and spear fishing and the collection of lobster and abalone. Best models, AICc values, weights and averaged coefficients are provided in supplementary material (Tables S1, S2)

for the biomass of targeted fish species, indicating that the effect of the protection level changed with aspect. Targeted fish biomass was higher in sheltered sites for fully protected (A) and unprotected sites (D), but lower in partially protected (B) and (C) sites. Cabbage Tree Bay Aquatic Reserve had a markedly higher biomass of targeted fish species than other MPA and non-MPA locations (Fig. 5).

The influence of Cabbage Tree Bay on these results was tested by removing this location. Whilst this resulted in a loss of significance, it did not substantially change the relative importance (RVI) of factors for richness. It did reduce the relative importance of protection level and increase the importance of habitat complexity for fish abundance and biomass, but aspect remained the most important factor.

Fish community composition

A total of 161 fish species were recorded across the surveyed sites; 81 large fish species (20 cm +) of which 17 were targeted fish species. A total of 62,290 fish individuals were recorded over all locations over the 3 years; 21,276 after exclusion of *A. strigatus, T. taeniatus* and *S. scalaripinnis*. A total of 1817 targeted fish were recorded, of which 1379 were legal size (legal sizes vary by species). Of all legal sized fish, 1240 (90%) were in no-take MPAs, 1188 of these (96%) were in the one MPA, Cabbage Tree Bay. The total fish biomass over all sites and years was 2527 kg of which 1816 kg was large fish and 905 kg was targeted fish.

Fig. 2 All fish (**a**), large fish (**b**) and targeted fish (**c**) species richness by significant factor aspect (wave exposure), for all surveys from 2015 to 2017, categorised by site protection level (PL); A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA
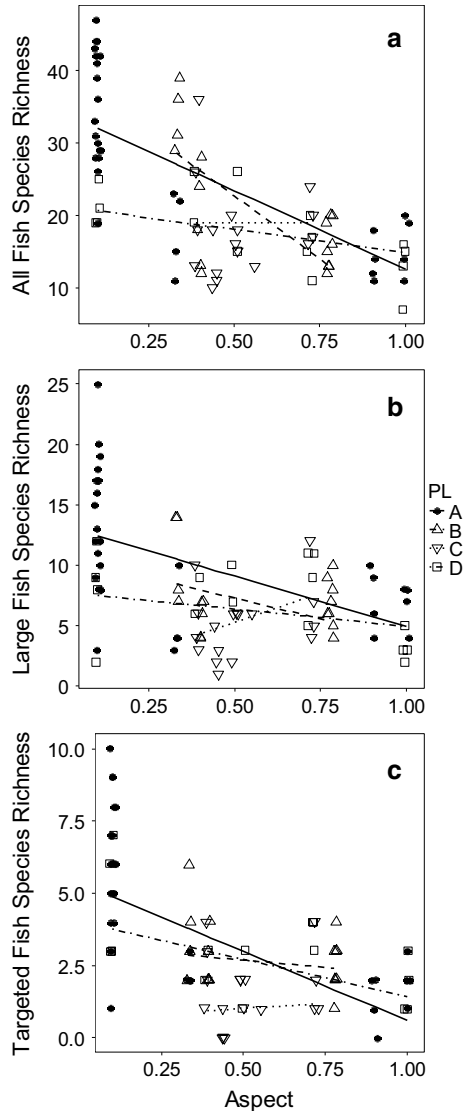
The most common fish species were crimson banded wrasse (*Notolabrus gymnogenis*) (detected on 98% of transects across all years), *T.taeniatus* (96%), white ear parma (*Parma microlepis*) (94%), and *A. strigatus* (89%). The most abundant fish species were *T. taeniatus* (total of 24,672 individuals across all transects and years), *A. strigatus* (12,105 individuals) and *S. scalaripinnis* (4237 individuals). The most abundant large fish were luderick (*Girella tricuspidata*) (844 individuals) and yellowtail scad (*Trachurus novaezelandiae*) (663 individuals), and the most abundant targeted fish were *G. tricuspidata* (845 individuals) and red morwong (*Cheilodactylus fuscus*) (240 individuals). Differences in large fish abundance between protection levels were related to eight fish species (Fig. 6).

Fig. 3 Large fish (**a**), targeted fish (**b**) and invertebrate abundance (**c**) by significant factors habitat complexity, aspect and algal canopy. Surveys from 2015 to 2017, categorised by site protection level; A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA
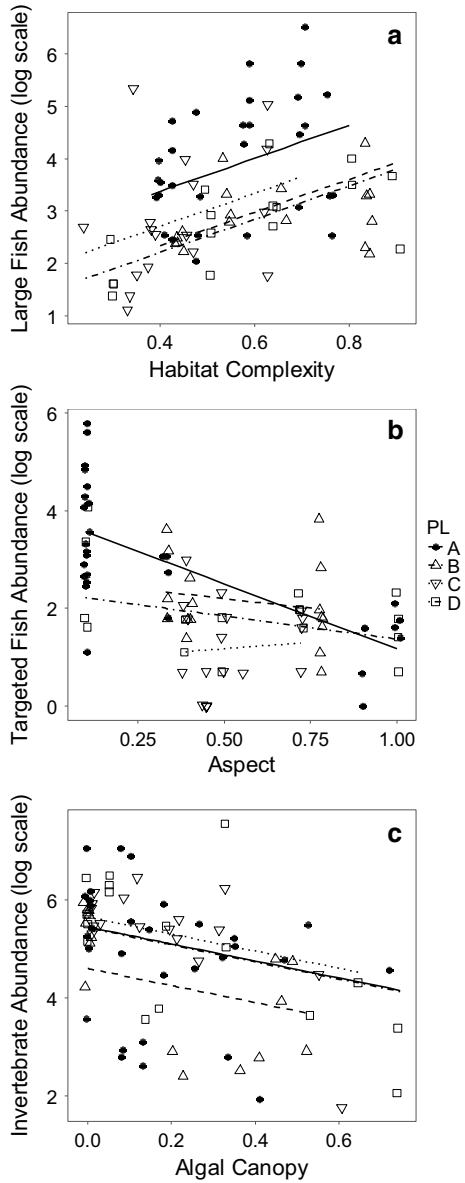

A number of species popular with recreational fishers were notable by their scarcity. Snapper (*C. auratus*) were scarce and small; only 11 individuals were recorded on transects in the 3 years, and all were undersize (< 30 cm). No mulloway (*Argyrosomus japonicas*) were recorded. Legal-sized mullet (*Myxus elongatus*) and tarwhine (*Rhabdosargus sarba*) were only recorded at Cabbage Tree Bay sites.

The highest fish biomass was contributed by *G. tricuspidata* (391 kg), blue groper (*Achoerodus viridis*) (343 kg) and *A. strigatus* (250 kg). *Girella tricuspidata* and *A.*

Fig. 4 All fish (**a**), large fish (**b**), and targeted fish (**c**) biomass by significant factors habitat complexity, aspect and protection level. Surveys from 2015 to 2017, categorised by site protection level; A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA
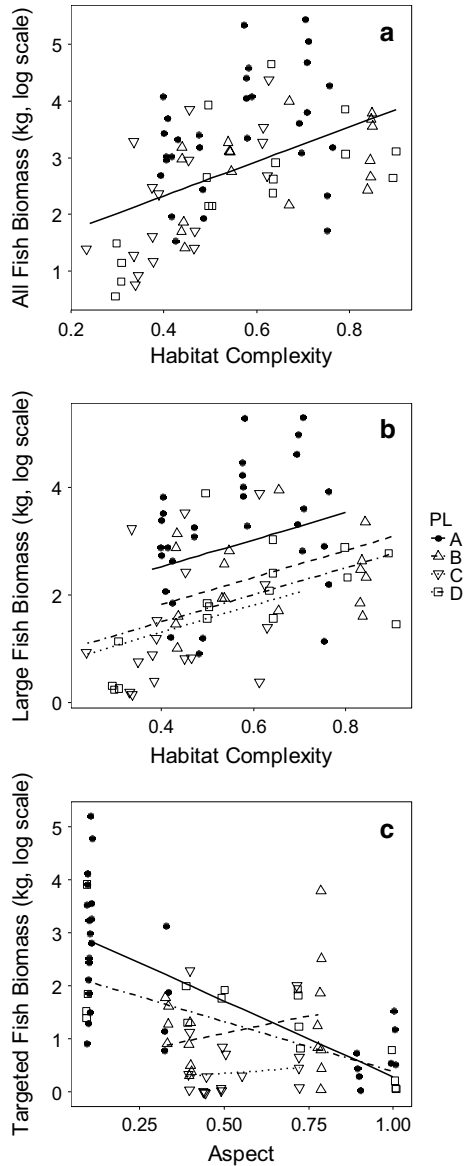
viridis also had the highest biomass of large and targeted fish species. A range of fish species, targeted and untargeted, contributed to differences in large fish abundance, where habitat complexity was the significant factor (Fig. 6). Diamondfish (*Monodactylus argenteus*), cornetfish (*Fistularia commersonii*), and longfin pike (*Dinolestes lewini*) are all untargeted species, and wobbegong (*Orectolobus maculatus*) may be targeted as a shark, but has a bag limit of zero. Differences in fish biomass across
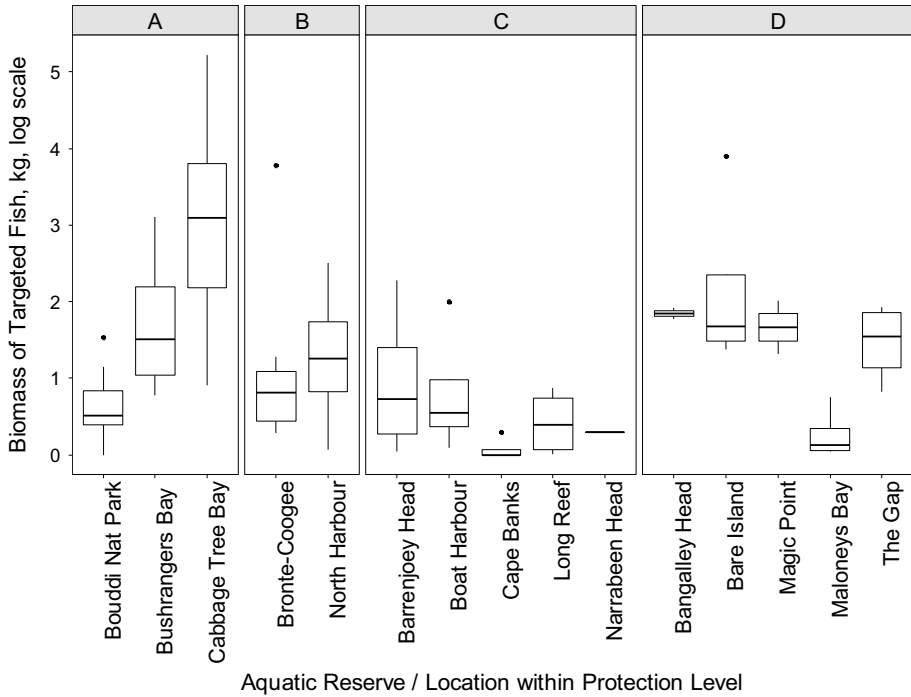
Fig. 5 Biomass of targeted fish species by study location (boxes indicate median, first and third quartiles and whiskers indicate values no further than 1.5×Inter-quartile range). A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA

protection levels were strongest for a number of targeted, unprotected fish species such as *G. tricuspidata* and *A. australis* (Fig. 7).

Mobile macro invertebrates

The most important variables in predicting mobile macro-invertebrate richness were aspect, protection level and the interaction between these, although none of these factors were significant. Invertebrates were significantly less abundant at sites with more algal canopy (Fig. 3). The RLS method requires large canopy-forming macroalgae to be parted and searched to record mobile macroinvertebrates living within and under the canopy, so this result is unlikely to be confounded. The effect of fish abundance on invertebrates was assessed and found to have lower importance (RVI) than other factors. This lack of effect may be due to the particular fish species which are more abundant in MPAs in the region—for example *G. tricuspidata* (which are herbivores) and *A. australis* (which are not large enough to feed on the most abundant invertebrate, *C. rodgersii*). Furthermore, one of the largest urchin predators—*A. viridis*—enjoys species-specific protection across all sites.
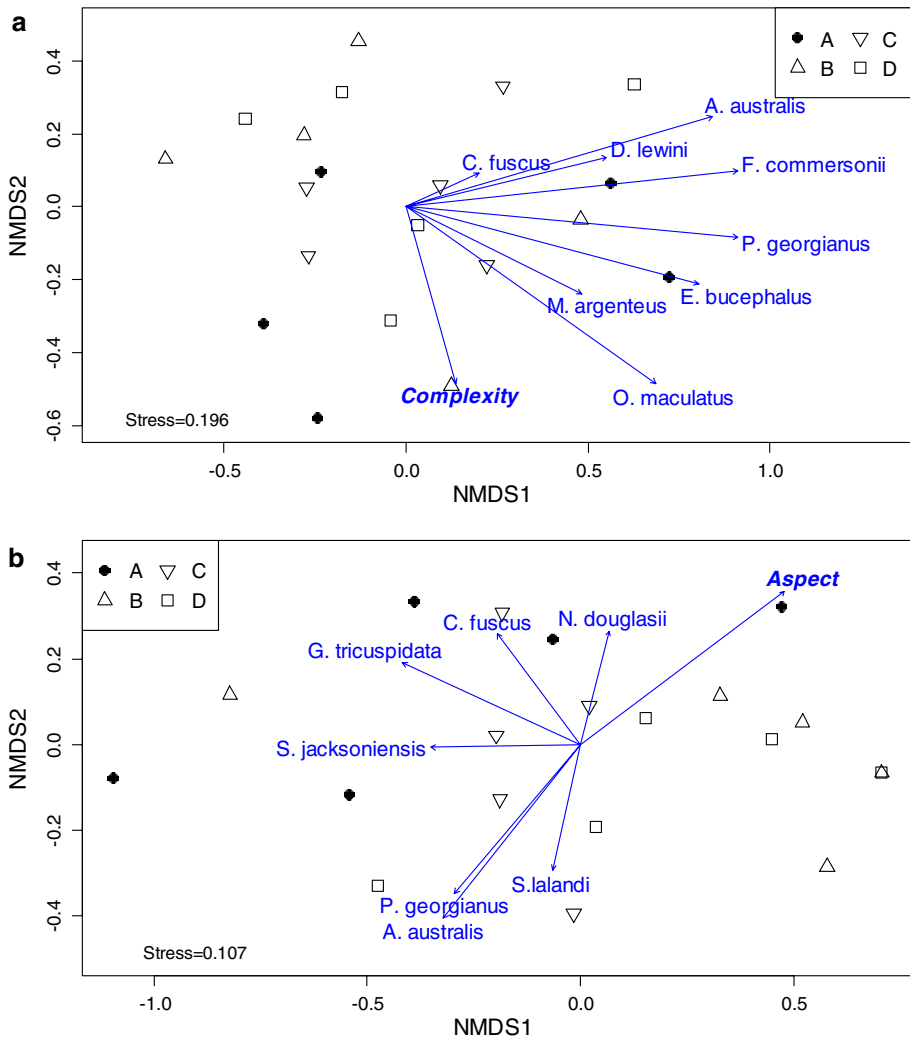
Fig. 6 nMDS ordination plot of similarity of large 20 cm + fish abundance (**a**) and targeted fish biomass (**b**) assemblages across 20 sites, categorised by site protection level; A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA. Vectors indicate species driving differences and the significant predictor variable (bold)

Community composition

A total of 53 invertebrate species and 20,930 invertebrate individuals were recorded over the 3 years of surveys. The most common invertebrate species were tent shells (*Astralium tentoriformis*) (present on 98% of transects), black urchins (*Centrostephanus rodgersii*) (88%) and turban shells (*Lunella torquate*) (51%). Figure 7 shows two abundant invertebrate species and algal canopy by protection level. These were also the most abundant
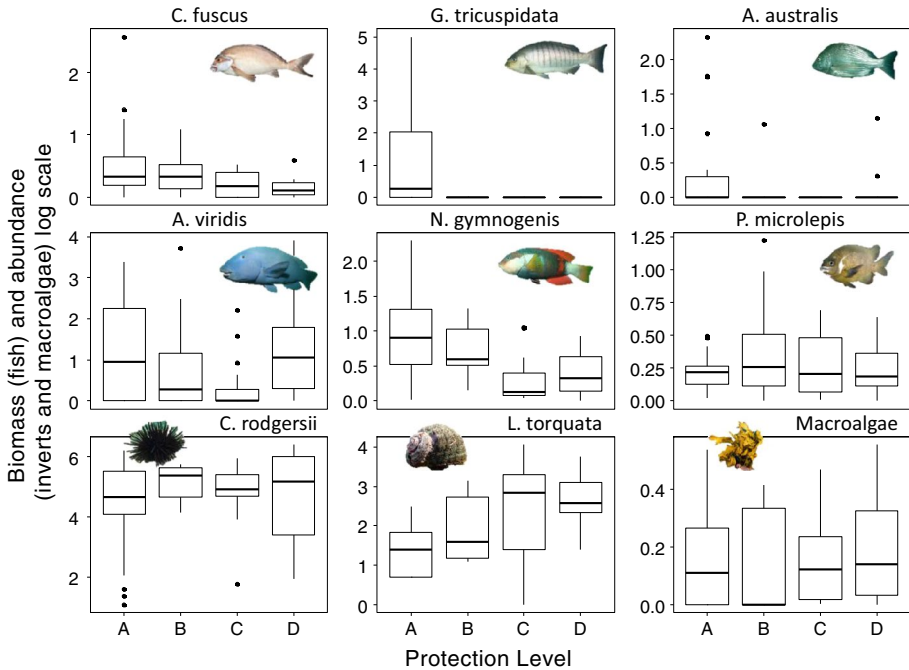
Fig. 7 Selected common species showing log biomass (fish), and abundance (invertebrates and large canopy forming macroalgae) per survey by protection level. The top row contains popular targeted species; the middle row contains non-targeted species (*N. gymnogenis* and *P. microlepis*) and *A. viridis* which enjoys species-level protection. Boxes indicate median, first and third quartiles and whiskers indicate values no further than 1.5×Inter-quartile range. Protection levels A—full no-take MPA; B—MPA allowing line fishing and collection of abalone and lobster only; C—MPA allowing line and spear fishing, and collecting lobster and abalone only; D—no MPA

species, with a total of 12,959 *C. rodgersii*, 6834 *A. tentoriformis* and 522 *L. torquata* across all years and transects. Differences in invertebrate abundance between protection levels were related to *L. torquata,* the tritons *Cabestana spengleri* and *Ranella australasia, C. rodgersii,* orange feather star (*Comanthus trichoptera*), slate pencil urchin (*Phyllacanthus parvispinus*) and the nudibranch *Doriprismatica atromarginata.*

Benthic communities

There were no strong patterns in the relative importance of predictor variables for benthic richness, and no significant differences between the benthic assemblages by protection levels. A total of 27 benthic morpho-taxa of sessile and mobile invertebrates and macroalgae were recorded. The highest cover taxa were large canopy-forming macroalgae and encrusting calcareous red algae (both 20% of cover). Large canopy forming macroalgae consisted predominantly of *Ecklonia radiata*; crayweed (*Phyllospora comosa*) was only recorded at the two most southern sites, Maloney's Bay and Bushranger's Bay. Urchin barrens, comprising urchins, encrusting algae and bare rock, represented 27% cover of the area studied.

Discussion

Given the original objectives of the MPAs, we define "effective" as having relatively high levels of biodiversity (species richness), abundance and biomass, compared to reference sites and/or MPAs with lower protection (Smith and Pollard 1996). In accordance with this definition, only no-take protection was effective for fish, and MPAs were not effective in protecting invertebrates, or benthic biota. The most important predictors driving fish communities were aspect, protection level, their interaction, and complexity. Invertebrate richness was related to the same factors (aspect, protection level, their interaction, and complexity), although none were significant, and invertebrate abundance fell significantly with increasing algal canopy. The benthic community was not significantly related to any of the predictors tested.

Of all the MPAs in this study, Cabbage Tree Bay was exceptional in terms of richness, abundance and biomass and appeared to be the main contributor to the significant differences between protection levels. Compared to other no-take MPAs it had twice the richness and abundance of all fish and large fish, four times the biomass of all fish and large fish and eleven times the biomass of targeted fish. This is in line with expectations according to the NEOLI criteria (Edgar et al. 2014), as Cabbage Tree Bay meets three of the five criteria (No-take, Enforced and Old). In addition to these criteria, we find that for small MPAs, exposure (indicated by aspect), complexity and algal canopy are important factors driving MPA effectiveness. The remainder of the discussion explores the potential mechanisms causing these results and provides recommendations for future management to improve the effectiveness of small MPAs in temperate rocky reefs.

Wind and wave exposure have been suggested as key physical factors influencing community structure in shallow aquatic habitats (Fulton and Bellwood 2004). Increased turbulence, current speeds and wind-induced upwelling bring nutrients to the MPAs; enhancing primary production in the form of micro-algal and benthic algal communities (Ballesteros 1989; Hepburn et al. 2007). Increased primary production can lead to an increase in invertebrate abundance, which can serve as food for different fish species (Sala 2004). On the other hand, too much exposure can also result in severe physical disturbance to benthic communities (Siddon and Witman 2003) and can limit primary production (Reed et al. 2011). In this study, aspect was used as a proxy for wind and wave exposure and it was one of the most important factors determining fish communities. Our results suggest that at the spatial scale of these small MPAs, low exposure has an important and positive effect on fish richness, abundance and biomass, particularly for targeted fish species when combined with no-take protection.

In both terrestrial and aquatic environments, structurally complex habitats are known to support a higher diversity and density of animals (e.g. Manatunge et al. 2000; Kovalenko et al. 2012; Harborne et al. 2011) as a greater range in habitats serve as shelter, reproduction sites and food resources (Charton and Ruzafa 1998; Johnson et al. 2003). The influence of complexity on fish, invertebrate and benthic biota has been recorded in both tropical (Harborne et al. 2011; Ferrari 2017), subtropical (Ferrari et al. 2017) and temperate (Rees et al. 2014) marine ecosystems. Recent studies have recognized that the role of structural complexity in influencing marine communities may depend on the focal species ecological and functional traits, such as trophic positioning, home range and body size (Harborne et al. 2012; Ferrari et al. 2016, 2017). Indeed, we found that complexity may play an important role in the abundance of large fish as well as the biomass of all fish and large fish. Our findings suggest that the recommendations made by Graham and Nash

(2013) regarding the need to quantify complexity of coral reefs in tropical MPAs, should be extended to temperate rocky reefs.

Structural complexity in temperate rocky reef systems can influence fish, invertebrate and benthic assemblages across multiple spatial extents and resolutions (Rees et al. 2014; Ferrari et al. 2017). Habitat structural complexity is the physical three-dimensional structure of an ecosystem, and it provides a variety of resources to organisms in marine ecosystems (see review by Graham and Nash 2013). Different attributes of complexity affect different biotic functional groups and in different directions, for instance herbivores may be attracted to less complex reefs while predators may prefer more complex reefs (Harborne et al. 2012; Ferrari et al. 2017). In reefs with high exposure to wave energy, complexity could also offer sheltered areas for organisms—this is particularly important in shallow (< 25 m) temperate reefs subject to high wave energy. Finally, complexity also influences reef fish behavior and thus important ecological processes; such as the distribution of planktivores, who prefer less complex areas where they can forage while monitoring their surroundings for predators (Rilov et al. 2007). While most of these relationships have been derived in tropical or subtropical reefs, it is possible that they apply to temperate reefs as well. To the best of our knowledge this is amongst the first studies to investigate the importance of complexity in predicting the richness, abundance and biomass of targeted fish species, and one of the first studies to look at the relationship between complexity and large fish richness, abundance and biomass in temperate reefs (but see Morris et al. 2018 for a review of the structural attributes that favour fish communities). Clearly, more studies investigating the relationship between complexity and fish in temperate rocky reefs are needed.

Protection level was significant for large fish richness and in interaction with aspect for the biomass of targeted fish species. The interaction was significant for no-take MPAs (Level A) protection versus lesser levels of protection. Partial protection was not significantly different to unprotected areas for any of our response variables. Differences in fish richness are often variable when comparing fully protected sites to unprotected sites at a continental scale (Edgar and Stuart-Smith 2009). Our study of small MPAs found significantly higher richness in no-take MPAs, but only for large fish. Increases in biomass, density and average body size are also common results of MPA studies (e.g. Lester et al. 2009; Edgar et al. 2014; Coleman et al. 2015) linked to exclusion of any kind of fishery. Given the tendency to declare MPAs with only partial protection to minimize negative social reactions (Roberts and Hawkins 2000; Edgar et al. 2004), our study indicates that such MPAs may be of little benefit to biodiversity, abundance or biomass. Our findings are supported by larger scale studies, which consistently identify 'no-take' classification as an essential attribute of a successful MPA (Costello and Ballantine 2015). MPAs may have additional social benefits, such as local education and awareness (Costello 2014), but we did not assess social benefits in this study.

Our study did not consider levels of fishing effort or accessibility of sites. Accessibility was generally well represented across the sites, for example low access/remote sites include Bouddi (A), Barrenjoey Head (C) and The Gap (D), and high access sites include Cabbage Tree (A), Bronte-Coogee (B), Long Reef (C) and Bare Island (D).

The scarcity of popular fished species such as snapper, mulloway, lobster and abalone; and the high proportions of undersize individuals across all fish agree with recent studies indicating high fishing pressure in the region (Ghosn et al. 2010). Historic accounts indicate that snapper was once harvested in the region in large numbers (Pepperell 2008). A recent study has shown that, when a small protected area is well-established, it is possible for mature individuals to contribute to settlement of juveniles outside the MPA area and

possibly benefit fisheries due to these spill-over effects (Le Port et al. 2017). Our study, however, notes the lack of not only mature snapper, but also a paucity of undersize individuals, possibly indicating declines in reproductive and recruitment processes for local populations of this species. Two popular fished invertebrates were also conspicuous by their scarcity; only three abalone (*Haliotis rubra*) and no lobster (*Jasus edwardsii* or *Sagmariasus verreauxi*) were recorded over the 3 years of surveys. Both abalone and lobster have active fisheries in NSW, and these results were in contrast to other studies which have reported the positive impact of MPAs on populations (Barrett et al. 2009). These results support evidence from other studies that found that poorly-designed and implemented MPAs combined with over-fishing can result in meta-population collapse (Hopf et al. 2016).

Eastern blue gropers are protected by law from spearfishing, and are rarely taken by recreational line fishers (Kingsford et al. 1991). The abundance of this species was not significantly different between unprotected areas and no-take MPAs. Abundance was well distributed across size classes, with a total of between 10 and 19 individuals in all size classes from 15 cm to 62.5 cm. This indicates that species-specific protection, rather than spatial protection may result in benefits for blue groper populations. Effective species-specific protection is important for signature species such as blue groper (Lee et al. 2015) and others such as weedy seadragons, which can also be important to gain public support for the designation of MPAs due to their charismatic nature and high site fidelity.

Algal canopy was the main factor associated with changes in invertebrate abundance and effects were species-specific. Abundances of invertebrates can be lower in MPAs (Edgar and Stuart-Smith 2009), possibly due to a trophic cascade when their predators are no longer being removed (Barrett et al. 2009). Some invertebrates, such as urchins (*C. rodgersii*) and turban shells (*L. torquata*) were less abundant in MPAs in our study, although the results were not significant. Several long-term studies have suggested a cascading effect on benthic communities following the removal of predators (Pinnegar et al. 2000). Results from the implementation of MPAs, however, demonstrate that algal densities may remain unchanged (Babcock et al. 2010) while invertebrate densities change. In this study there was no significant difference in benthic communities even when effects on potential predators (fish) were observed, possibly due to the small size of the MPAs that allows fast recruitment from nearby areas. Urchin barrens were more extensive than large canopy-forming algae in total across the sites studied (27% vs 20%) but the distribution of barrens was not related to protection levels. An increase in urchin densities is often related to the over-exploitation of large predators, that are the primary consumers of urchins, leading to the formation and persistence of barren grounds (Pinnegar et al. 2000; Shears and Babcock 2002). Kelp and other macroalgal abundance could be negatively influenced by high densities of sea urchins by overgrazing (Carnell and Keough 2016), and have been studied on temperate rocky reefs worldwide (Petraitis and Dudgeon 2004; Ling 2008; Filbee-Dexter and Scheibling 2014). They are considered alternative stable community states with several mechanisms causing the change from one state to the other (Estes et al. 1998; Konar 2000; Lafferty 2004). In our study, species known to prey on urchins have a generally low abundance (e.g. snapper and lobsters) or are protected from fishing both inside and outside MPAs (e.g. blue groper). This may explain the lack of variation in invertebrate and benthic communities across all MPAs.

All the MPAs in the Hawkesbury shelf bioregion can be considered small and relatively old (Edgar et al. 2014) and neither size nor age were found to be significant in our analysis. This is despite the fact that some MPAs sampled in this study are 15 years old and changes in species abundance or biomass are known to take up to 20 years to

manifest (e.g. Edgar and Stuart-Smith 2009; Babcock et al. 2010). The lack of significance of size and age in our study may be explained by the absence of large or young MPAs in the bioregion for comparison. Size is generally considered to be one of the most important factors in protected areas, with larger MPAs and coordinated networks of MPAs shown to be more effective (Edgar et al. 2014; Costello et al. 2010). Large-scale MPAs and networks can be comprehensive and adequate to protect biodiversity and can represent all habitats in a bioregion (Fernandes et al. 2005). But it may be difficult to establish MPAs on such scales for social and political reasons (Agardy et al. 2003). Whilst small MPAs are not a substitute for large-scale MPAs or networks, our study shows they can be effective at a local scale. We find that placing a small no-take MPA in a sheltered location with complex habitat results in higher effectiveness in terms of biomass and biodiversity. Should other criteria be important, such as protecting a threatened species, a unique community or spawning area, then other locations may be suitable.

The limited home ranges of some species are thought to be protected within MPAs of < 0.5–1 km across (Taylor and Mills 2013; Green et al. 2014). Although this range is not sufficient for many other species, protecting a location that is valuable for spawning, nursery or aggregation of these species might be sufficient to generate a spill over to adjacent areas (Green et al. 2014; Gaines et al. 2010). However, if an MPA is too small, biomass within the MPA may be low due to a large spill over rate (Kramer and Chapman 1999; Botsford et al. 2003; Gaines et al. 2010). While we did not explicitly study spill over or boundary effects, there is no indication in our study that locations near Cabbage Tree Bay, such as North Harbour and Long Reef, were benefitting from spill over. Boundary effects such as "fishing the line" may also be important (Kellner et al. 2007), particularly in small MPAs. We did not observe any visible evidence of this phenomenon during our surveys.

The effectiveness of Cabbage Tree Bay is not surprising since this is the only MPA that meets with the minimum of three NEOLI criteria as proposed by Edgar et al. (2014); it is No-take (N) and Old (O) but additionally it appears to meet another criterion, being well Enforced (E). Community support and compliance, and enforcement are important considerations for effective MPAs (Kareiva 2006; Kelaher et al. 2014; Camargo et al. 2009). The current zoning regulations are extremely complicated and varied throughout the bioregion, and more complex rules make compliance and policing more difficult (Costello and Ballantine 2015). We did not design our study to investigate enforcement and assumed a relatively consistent level of formal enforcement across sites on the basis that they are covered by a single enforcement body (NSW Department of Primary Industries) and are in a single bioregion. We observed no formal enforcement activities, such as patrols, in any MPAs during our surveys. We did observe non-compliance (line fishing and taking of octopus) in two no-take MPAs, Bushrangers Bay and Bouddi, but no such activities in Cabbage Tree Bay. Bushrangers Bay and Bouddi are more remote and in less populated areas, with no visible, informal enforcement by the local community. The Cabbage Tree Bay Aquatic Reserve is noteworthy in being no-take, situated in a highly populated, visible location and having an active local community including a 'Friends of Cabbage Tree Bay' group, who are vigilant in identifying non-compliance and actively communicate MPA restrictions (MEC 2013). Local stewardship has been identified as critical to success of marine spatial planning (Domínguez-Tejo et al. 2016) and it may be that it is an important social factor contributing to the success of this MPA. Several other MPAs, such as Long Reef and Bronte-Coogee, have active and supportive local communities however the effect of any informal enforcement in these MPAs is limited by the narrow scope of their partial protection. Enforcement in many of the bioregion's MPAs is likely to have little ecological

benefit as they restrict little extractive activity—for example by protecting invertebrates yet allowing two desirable species—abalone and lobster—to be taken.

Conclusion

We conclude that MPAs, as currently designated in this bioregion, are generally not effective in achieving their original goals; conservation of biodiversity (species richness) and improved fish stocks (abundance and biomass) (Smith and Pollard 1996). Whilst all MPAs have been in place for 15 years or more, none are large or isolated, so they can achieve at best three of the five NEOLI criteria—No-take, Enforced and Old (Edgar et al. 2014). Seven out of the ten MPAs have only partial protection, and these were not significantly
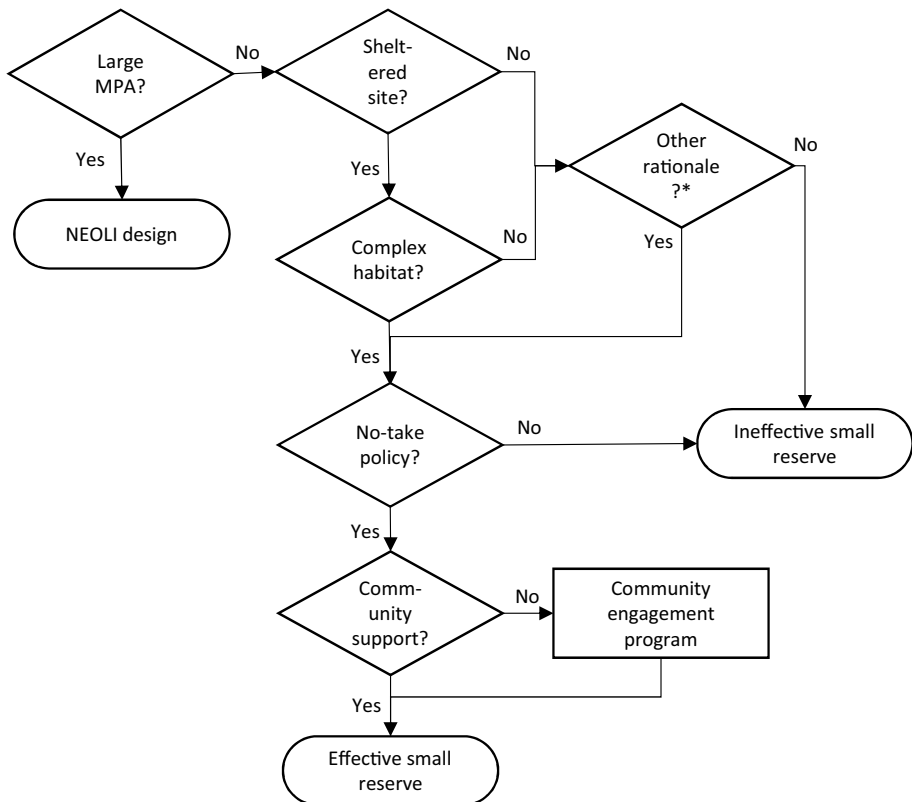
Fig. 8 Decision framework for temperate MPAs. Symbols indicate: diamond = decision, square rectangle = process, rounded rectangle = conclusion. Large MPAs exceed 100 km². Sheltered sites have low wave exposure, facing away from prevailing major storm/swell direction. Complex habitat includes walls, boulders and overhangs. Other rationales (other than biodiversity and biomass conservation) may exist for declaration of an MPA, such as local education opportunities, meeting Comprehensive, Adequate and Representative (CAR) requirements in the context of other MPAs in the region and protection of threatened species or key spawning areas. If these are part of the rationale, this should be made clear to ensure monitoring and expectations of performance are properly aligned

different from unprotected areas for any of our response variables. Of the three no-take MPAs, one MPA—Cabbage Tree Bay—appears to account for the majority of the positive MPA effects, and this location is naturally sheltered, has a diversity of habitats and benefits from informal enforcement of protection by the local community. At 0.2 km^2, this MPA makes up just 0.01% of the coastal waters of the bioregion (MEMA 2016) and its effects appear to be highly localised.

When combined with our other findings, this suggests that small MPAs can enhance biodiversity and biomass at a local scale, but only if they have full no-take protection, are in sheltered locations with complex habitat, and have positive community involvement to engender support and stewardship (Costello 2014). Given these findings, it would be wise to conduct a spatial analysis of the bioregion to identify locations which meet these criteria, using the decision framework in Fig. 8.

These results provide a baseline for robust assessment of the effectiveness of small MPAs and inform future management decisions and small MPA design in other locations.

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