

VEAC Marine Investigation

Resilience of Victorian reefs to climate change:

An investigation utilising the Sub-tidal Reef Monitoring Program

Carnell, P.E. and Longmore, A.R.

April 2014



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Executive Summary

Victoria's marine protected areas (MPAs) include "no-take" and "multiple-use" areas. The primary ecological purpose of the no-take areas is to maintain examples of Victoria's biodiversity. The intention is to preserve areas of the marine environment in relatively natural condition for their intrinsic value and to benefit future generations. While the "multiple-use" marine parks, marine reserves and marine and coastal parks were created to protect significant ecological values, they are also intended to be managed in a way that accommodates sustainable use of resources including, but not limited to, fishing. Victoria's marine environment appears to be changing as a result of changing climate. The potential effects of Victoria's changing climate on the biodiversity of the MPAs were not well recognised when the areas were established, and their implications to the MPA system have not been formally considered.

It is possible that the MPAs may be more resistant or more resilient than other marine areas to the disturbances that are likely to result from climate change. Resistance is defined as the ability of an ecosystem to withstand change despite forces trying to push it one way or the other, while resilience is the ability of the ecosystem to recover and return to its previous state following a disturbance that may result from such a force.

One immediate threat potentially facilitated by a changing climate is to abalone (*Haliotis rubra*) habitat on eastern Victorian reefs. This area has seen the incursion of the longspine sea urchin (*Centrostephanus rodgersii*), which has spread from NSW as far west as Wilsons Promontory, possibly assisted by the southward extension of the East Australian Current over the past 50 years. The urchin eats macroalgae preferred by abalone, creating barren areas which can no longer support abalone. Ecological theory suggests that the more "natural" an MPA is, the more resilient it will be to disturbance, compared to nearby unprotected areas. In Tasmania, overfishing of a key urchin predator, rock lobster, has also aided urchin expansion and where abundances of 'large' lobsters are sufficient, urchin numbers are kept low. Victorian MPAs currently affected by *Centrostephanus* include Wilsons Promontory, Point Hicks and Cape Howe Marine National Parks and Beware Reef Marine Sanctuary.

Victoria has available a long time series of monitoring data that describes aspects of the biodiversity of sub-tidal reefs in a subset of Victoria's no-take MPAs. These data, collected through the Sub-tidal Reef Monitoring Program (SRMP) conducted by Parks Victoria, have been obtained from regular surveys at locations throughout Victoria, encompassing both representative and unique habitats and communities (Edmunds and Hart 2003). The Scientific Advisory Committee for the VEAC Investigation suggested that a targeted analysis of these data could explore the question of whether Victoria's MPAs appear to have higher resilience than surrounding marine areas to the disturbances associated with the range extension of *Centrostephanus*. The SRMP was not designed to address this specific question, and it is therefore possible that the indicators and sampling approach may limit its effectiveness for this purpose. The design of the SRMP does, however, include the spatial and temporal components (to varying degrees) required for a quantitative assessment. The analysis also has potential to serve as a case study demonstrating how this dataset can be applied to explore such specific questions to inform management, research or future refinement of the monitoring program.

We therefore aimed to:

1. Determine if the existing sub-tidal reef monitoring data is capable of supporting a robust statistical assessment of whether MPAs have higher resilience to this threat than reference areas outside the MPAs.
2. Identify those Victorian MPAs where *Centrostephanus* is present and ascertain, to the extent possible using the existing sub-tidal reef monitoring data, the current relationships between *Centrostephanus*, habitat-forming algae and *Haliotis rubra* (black-lip abalone).
3. Compare, statistically, the abundances (and where applicable sizes) of *Centrostephanus*, habitat-forming algae and other key reef species in the MPAs to comparable areas outside the MPAs (for those MPAs where *Centrostephanus* is present).
4. Assess whether the existing monitoring data is fit for this purpose and suggest any appropriate refinements to the program in this context.

A range of analyses of SRMP data were carried out based on recommendations from earlier reports, where the type of analysis to be used depends on size of MPAs, suitable reference sites outside MPAs and whether surveys have been conducted before and/or after declaration of the MPA system. We concluded that the SRMP data can be used to make an assessment of differences inside compared to outside MPAs. We tested specific hypotheses relating to how climate change may affect resilience and resistance to change of a range of species inside compared to outside MPAs. While we could examine the relationship between *Centrostephanus* and various reef species, this was not possible in a statistical sense (due to the survey design) or in relation to MPA protection (due to low replication). However, building upon these relationships we could then test if there is a difference in the number (or where applicable size) of species inside compared to outside. If the information from the analysis indicated an increase in lobster, decrease in *Centrostephanus* and increase in habitat-forming seaweeds in MPAs compared to outside, this could indicate greater resistance or resilience of MPAs to this threat. The specific hypotheses tested in relation to resistance or resilience included:

Resistance:

- *Centrostephanus* numbers increased or are in greater abundance outside compared to inside an MPA.
- Concordantly, abundance or size of other key reef species would decrease or be lower outside compared to inside an MPA.

Resilience:

- Depending on state of 'invasion', *Centrostephanus* numbers would decrease in abundance inside compared to outside an MPA if similarly highly abundant at the beginning of monitoring.
- Concordantly, abundance or size of other key reef species would increase inside compared to outside an MPA.

Resilience of MPAs is best tested with a long, comprehensive dataset, whereas resistance is able to be determined from a shorter time series. *Centrostephanus rodgersii* abundance differed between areas, with the highest abundances at Cape Howe and the lowest at Wilsons Promontory. Where *Centrostephanus* abundances were high at Cape Howe, there was a correspondingly low abundance of abalone and habitat-forming algae. These negative relationships were not observed in the other locations.

There was no clear difference inside and outside MPAs in relation to *Centrostephanus* abundance and associated declines in other species. However, at Pt Hicks MNP abalone were larger overall within the MPA and abalone were more abundant in Cape Howe MNP when compared to reference areas. In Wilsons Promontory MNP, we also saw a small increase in overall *Jasus edwardsii*

(rock lobster) abundances inside the MPA compared to a decrease outside since the MPA was declared. Given that it is the abundance of 'large' individuals of *Jasus edwardsii* that provides resistance/protection for a reef to *Centrostephanus*, the abundance of this species is paramount to promoting reef resilience in this context.

While the SRMP is capable of detecting changes for common species, statistical power to detect change is low because of the high variance in most species. Where threats relating to less common species are of concern, specific survey techniques are needed to improve the ability to detect meaningful changes. For the MPAs considered, this could include additional methods specifically for *Centrostephanus* and *Jasus edwardsii*, and other ecosystem components such as canopy algae, where they may contribute to resilience.

Introduction

The coastal and marine environments of Victoria include bays, inlets and estuaries, Bass Strait and the open ocean. Victoria is responsible for about 10,600 km² of marine waters, including those from the coast to the three nautical mile limit (about 7,000 km²), and a further 3,600 km² in the coastal bays (Port Phillip, Western Port and Corner Inlet). These waters contain many species of marine flora and fauna found only in southern or south-eastern Australia. The increasing movement of people toward coastal areas inevitably increases pressure on natural resources near the coast. The Environment Conservation Council carried out an investigation of Victoria's marine, coastal and estuarine areas (ECC 2000), the purpose of which was to make recommendations on the protection of areas with significant environmental values and the sustainable use of these areas.

This investigation led to the development of a system of no-take Marine Protected Areas (MPAs) in 2002, the primary objective of which is to protect biodiversity and to preserve representative examples of natural ecosystems. This system includes 13 no-take marine national parks and 11 marine sanctuaries established under the National Parks Act 1975, which cover 5.3% of Victoria's state waters. Victoria also has six "multiple-use" marine parks, marine reserves or marine and coastal park areas which were established prior to the no-take system and cover 6.4% of Victoria's state waters. While these multiple-use MPAs were created to protect significant ecological values, they are intended to be managed in a way that accommodates sustainable use of resources including, but not limited to, fishing. There has been no formal assessment of whether the MPAs are meeting the purposes for which they were intended when established more than a decade ago. Further, the potential effects of Victoria's changing climate on the biodiversity of the MPAs were not well recognised when the areas were established.

The Minister for Environment and Climate Change has requested the Victorian Environmental Assessment Council (VEAC) to carry out an investigation into the outcomes of the establishment of Victoria's existing marine protected areas. The investigation requires VEAC, an independent council, to assess how Victoria's marine protected areas have performed and how effectively they are being managed, in relation to the purposes for which they were established – particularly the ecological purposes. VEAC will also identify and assess the threats and challenges the areas may face in the future.

The purpose of the marine investigation is to examine and provide assessment of:

- (a) the performance and management of existing marine protected areas in meeting the purposes for which they were established, particularly the protection of the natural environment, indigenous flora and fauna and other natural and historic values; and*
- (b) any ongoing threats or challenges to the effective management of existing marine protected areas, particularly in relation to the biodiversity and ecological outcomes.*

The project reported here will inform VEAC's assessment for components (a) and (b) of the investigation. The specific objectives of the project are discussed in the sections below.

Ecological context for investigation (component b)

Climate change, including increasing temperature, increasing acidification and changes in frequency and intensity of extreme weather events, is predicted to have a range of effects on marine species (Byrnes *et al.* 2011, Klemke and Arundel 2013, Smale and Wernberg 2013). One of the key processes expected to drive changes in these physiochemical parameters in south-eastern Australia is the strengthening of the East Australia Current, which transports warm, saline water down Australia's eastern coastline shedding eddies into eastern Bass Strait (Klemke and Arundel 2013). Given that the average yearly sea-surface temperature in south-eastern Australia has been above the long-term average since the 1960s, some of

these changes have been happening gradually for considerable time (Johnson *et al.* 2005, Cleugh *et al.* 2011, Wernberg *et al.* 2011). Recent research is now revealing climate change impacts on marine species and communities which in some cases have had dramatic ecological consequences (Johnson *et al.* 2011, Wernberg *et al.* 2013).

The range expansion of *Centrostephanus rodgersii*, the longspine sea urchin (hereafter *Centrostephanus*), into Bass Strait and eastern Tasmania has had dramatic effects on reef ecosystems over the last 20 years (Johnson *et al.* 2011). These effects result primarily from high densities of *Centrostephanus* destructively over-grazing kelps and other brown algae that are important for forming habitat on reefs. The removal of habitat-forming macroalgal species can have severe flow-on effects on both reef biodiversity (Ling 2008) and fished species (Johnson *et al.* 2011, Strain and Johnson 2012, Strain *et al.* 2013). Thus, the potential for *Centrostephanus* to pose both ecological and economic threats in areas that it colonises is high (Johnson *et al.* 2005). While such impacts have been documented in Tasmania, *Centrostephanus* has also been observed in recent years on various reefs in eastern Victoria. How the urchin may be influencing Victorian reefs and Marine Protected Areas is relatively unknown.

Ecosystems that are in more natural condition are predicted by ecological theory to be generally more resilient to disturbances (Naeem and Li 1997, Tilman *et al.* 2006, Doumin *et al.* 2012). This prediction may apply to the disturbances expected to result from climate change, including disturbances caused by species range expansions like that of *Centrostephanus*. Some marine case studies have now demonstrated this effect, including one involving increased capacity of some MPAs in Tasmania to resist these destructive impacts from *Centrostephanus*. This capacity has been attributed to greater abundance and size of rock lobster predators inside MPAs (Ling and Johnson 2012). Thus, the 'resistance' or 'resilience' of those reefs (MPA or otherwise) to over-grazing by *Centrostephanus* comes from greater abundance and size (carapace length >140 mm) of their predators. Resistance is defined as the ability of an ecosystem to withstand change despite forces (eg. *Centrostephanus*) trying to push it one way or the other, while resilience is the ability of the ecosystem to recover and return to its previous state following a disturbance that may result from such a force (eg. creation of a *Centrostephanus* barren and then recovery).

While this effect has been observed in Tasmania, it is not clear whether Victorian MPAs are likely to be more resistant or resilient than surrounding marine areas to over-grazing by *Centrostephanus* as has been observed for MPAs in Tasmania. Victoria's MPAs may differ from those in Tasmania in a variety of ways including their ecological nature, starting condition, ecological objectives, fishing history and management history. There may be differences in the top-down (high-order predator driven) or bottom-up (environmentally driven) forcing of ecosystem dynamics. One or more of these differences could affect the qualities that promote resistance and resilience to disturbances such as urchin over-grazing. Fairweather (2012) emphasises that marine protected areas have been established and managed to achieve at least two objectives in biodiversity protection:

- **an insurance role:** whereby protection is provided now for future benefits by reserving areas that are currently in good condition, essentially trying to keep them as pristine as possible
- **a remedial role:** whereby protection is provided to allow some areas to recover from a degraded state by protection now and into the future to remove past problems from human activities.

Changes in management such as establishment of MPAs can influence top-down dynamics. The type of ecological effects that are expected to result from protection depend on which of these objectives applies to a given MPA (Fairweather 2012).

Victoria has available a long time series of monitoring data that describes aspects of the biodiversity of sub-tidal reefs in a subset of Victoria's no-take MPAs. These data, the Sub-tidal Reef Monitoring Program (SRMP) established by Parks Victoria, have been obtained from regular surveys at locations throughout Victoria, encompassing both representative and unique habitats and communities

(Edmunds and Hart 2003). Such a large, comprehensive, publically funded dataset is relatively rare and has yielded vast amounts of information on a variety of fish, invertebrate and algal species in Victorian waters (Edmunds *et al.* 2003, 2005, 2007, 2010, 2012, Williams *et al.* 2007). However, this data series has only been analysed once relatively early on in its time series using appropriate quantitative statistics (Keough and Carnell 2009). The Scientific Advisory Committee for the VEAC Investigation suggested that one question that could be explored through a targeted analysis of these data is whether Victoria's MPAs appear to have higher resilience than surrounding marine areas to the disturbance associated with the range extension of *Centrostephanus*. The SRMP was not designed to address this specific question, and it is therefore possible that the indicators and sampling approach may limit its effectiveness for this purpose. However, the analysis also has potential to serve as a case-study demonstrating how this dataset can be applied to explore such specific questions to inform management, research or future refinement of the monitoring program. It is in this context that the current study has been commissioned.

Study objectives

Assessing whether MPAs respond differently to a threat (eg. climate change) than areas outside MPAs from existing monitoring data involves three main requirements, which were laid out by Keough and Carnell (2009):

1. A specific research aim or question associated with each MPA, against which performance can be assessed. This aim must be translated into a formal scientific target (e.g. threshold) in order for techniques such as power analysis to be used.
2. A sampling design that will allow the separation of effects of the MPA from other factors that influence biodiversity.
3. Data with statistical properties that will allow confident statistical decisions to be made.

If data analysis does not demonstrate an "effect" of a particular MPA, there are several potential explanations:

- a) It may be that the kinds of threats against which an MPA provides protection (e.g. climate change) are not severe in the particular MPA over the time period covered by the data. Such an outcome does not detract from the value of an MPA as insurance against future threats and may not be unexpected if the MPA was in relatively good condition when it was established. Such an "insurance purpose" is, in fact, one of the major purposes for which MPAs are established (see Fairweather 2012) and aligns with the purpose for which Victoria's MPAs were established (see Introduction above and VEAC 2012).
- b) Insufficient time might have elapsed for such effects to occur (e.g. if the population of a species was very low in an area prior to declaration, increases in numbers of this species may require considerable time).
- c) The amount of variation in the data might limit our ability to detect changes of a relevant size.

With this in mind, the overall objective of the project reported here was *to examine whether data from existing monitoring programs could provide evidence of resistance/resilience of Victorian MPAs to the effects of the introduction of a new species whose range is expanding as a result of climate change.*

Its more specific objectives are to:

1. Determine if the existing sub-tidal reef monitoring data is capable of supporting a robust statistical assessment of whether MPAs have higher resilience or resistance to this threat than reference areas outside the MPAs.
2. Identify those Victorian MPAs where *Centrostephanus* is present and ascertain, to the extent possible using the existing sub-tidal reef monitoring data, the current relationships between *Centrostephanus*, habitat-forming algae and *Haliotis rubra* (black-lip abalone).
3. Compare, statistically, the abundances (and where applicable sizes) of *Centrostephanus*, habitat-forming algae and other key reef species in the MPAs to comparable areas outside the MPAs (for those MPAs where *Centrostephanus* is present).
4. Assess whether the existing monitoring data is fit for this purpose and suggest any appropriate refinements to the program in this context

Methodology

Overview of the Sub-tidal Reef Monitoring Program

The Sub-tidal Reef Monitoring Program (hereafter SRMP) was established prior to MPA declaration in 2002. Due to financial and logistical constraints in conducting the SRMP surveys, the MPAs were prioritised for surveys both before and after MPA declaration. Thus, some MPAs have an extensive dataset, while others have been surveyed less frequently. The characteristics of the data available from each MPA are listed in Table 1. The abundance of fish and invertebrates, and percentage cover of algae are measured in the SRMP. Data collected in the SRMP comes from one 200 m long transect for each sampling site, generally running along the 5 m depth contour, with data recorded from smaller units within that transect. These include 4 x 50 m strips 10 m wide (= 2000 m²) for mobile fish, 4 x 50 m strips 1 m wide (=200 m²) for cryptic fish and invertebrates, and 20 x 0.25 m² quadrats (=5 m²) for percentage cover of macroalgae and sessile invertebrates (Edmunds and Hart 2003). Keough *et al.* (2007) argued that the smaller 50 m strips within the single transect sampled at each site are not independent of each other, and therefore should not be treated as replicates in statistical analysis. For this report, the data were therefore aggregated from the original data files to provide a single measure for each site-survey combination before they were analysed. Where appropriate, abundances were log-transformed before analysis, to make the underlying statistical distributions closer to normality.

The number of sites inside and outside the MPA differs between MPAs. Sites were located based on available sub-tidal reef. Two sites (one inside, one outside) were located at Beware Reef and eight (four inside, four outside) at Point Hicks. Eight sites (four inside, four outside) were also located at Cape Howe, but the locations of two outside and one inside the MPA were changed in 2009. At Wilsons Promontory, 28 sites were evenly distributed inside and outside the MPAs.

The hypotheses we tested in the following analyses were based on the outcomes of studies in NSW (Hill *et al.* 2003) and Tasmania (Ling 2008; Johnson *et al.* 2011; Strain *et al.* 2013). While we could examine the relationship between *Centrostephanus* and various reef species, this was not possible in a statistical sense (due to the survey design) or in relation to MPA protection (due to low replication). However, building upon these relationships we could then test if there was a difference in the number (or where applicable size) of species inside compared to outside. If the information from the analysis indicated an increase in lobster, decrease in *Centrostephanus* and increase habitat-forming seaweeds in MPAs compared out outside, this could indicate greater resistance or resilience of MPAs. The specific hypotheses tested in relation to resistance or resilience included:

Resistance:

- *Centrostephanus* numbers increased or are in greater abundance outside compared to inside an MPA.
- Concordantly, abundance or size of other key reef species would decrease or be lower outside compared to inside an MPA.

Resilience:

- Depending on state of 'invasion', *Centrostephanus* numbers would decrease in abundance inside compared to outside an MPA if similarly highly abundant at the beginning of monitoring.
- Concordantly, abundance or size of other key reef species would increase inside compared to outside an MPA.

Objective 1

In recent reviews of the design and initial data output of Parks Victoria's Sub-tidal Reef Monitoring Program, Keough *et al.* (2007) and Keough and Carnell (2009) described several approaches that could be used in monitoring program design and analysis to provide confidence that any biological changes or patterns that may occur in an MPA are due to MPA declaration and management, as distinct from, for example, changes due to natural temporal variation or other causes. We can be most confident of attributing change to an effect of an MPA when there is as much detailed information about the state of an MPA before and after declaration (or any other shift in management), with increased confidence provided by the use of reference sites, ideally with replication. For the Victorian MPAs, the extent of monitoring varies dramatically between sites, depending on factors such as the period over which a particular area has been monitored and the physical local conditions, which may restrict monitoring efforts.

In the absence of before-declaration data, or when the after-declaration data extends for much longer than the before-declaration data, we can alternatively compare the trends since declaration in sites inside and outside MPAs through time (e.g., Edgar and Barrett 1999; Russ and Alcala 2003; Micheli *et al.* 2004; Williamson *et al.* 2004). In these sampling designs, confidence in the conclusions that can be drawn from analysis of the monitoring data typically rises dramatically with sampling of additional sites, and from longer time series. If an MPA is small, or there are few comparable reference sites surrounding it, as is the case for several of Victoria's Marine Sanctuaries, a shift in focus from sampling multiple sites to generating long time series was recommended by Keough and Carnell (2009). The statistical analysis to be used for these reserves would become more robust as the length of the time series increased.

For objective 1, data collected in the SRMP were examined to see how well they complied with the requirements described above for robust analysis for the current purpose. The primary properties of importance for analysis of the SRMP are presented for each MPA in Table 1. Additionally, we also required predictions to which we could then assess the 'resistance' and/or 'resilience'. The 'resistance' of an MPA can be assessed based on one or few surveys (confidence of resistance comes from multiple surveys), where a difference in the abundance of a 'threat' outside compared to inside an MPA would suggest resistance (Figure 1). Whereas, assessing for 'greater resilience' would typically require longer time series, looking for convergence of time series as a threat arrives in an area, and then divergence in the MPA as it recovers while outside the MPA remains stable (Figure 1). A sampling protocol incorporating seasonal and year-to-year variation would be better suited to picking up these kinds of changes.

Objective 2

The data from the SRMP were used to identify the MPAs in which *Centrostephanus* occurs. As part of the SRMP, *Centrostephanus* has been recorded in the four most eastern Victorian MPAs (Figure 2 Figure 2a, b). From east to west these are: Cape Howe MNP, Pt Hicks MNP, Beware Reef MS and Wilsons Promontory MNP. To get an initial understanding of *Centrostephanus* numbers at the different parks, we looked only at surveys that had been conducted in these MPAs *after* declaration. We averaged across surveys at sites to calculate an average value per site; these values were used as the replicate in the visual comparison of numbers of *Centrostephanus* inside and outside at the four MPAs (Figure 2). No data analysis was run for this comparison.

We also plotted the relationships between abundance of *Centrostephanus*, habitat-forming algae and *Haliotis rubra* (black-lip abalone) to provide an indication, from these correlative data, of

how *Centrostephanus* may be affecting these species regardless of protection. As for the previous comparison these plots only used data from each survey conducted since 2002 (i.e. after declaration). We translated the *Centrostephanus* count on each transect to a density value (m^{-2}) by dividing the total number by the area covered (200 m^2). This allows for better comparison to other studies that typically use this measure of *Centrostephanus* abundance. We plotted these relationships separately for Cape Howe, Pt Hicks and Beware Reef. Wilsons Promontory was not included as the densities of *Centrostephanus* encountered there were so low. A line of best-fit and R^2 value are presented in the graphs.

Given the range in size of the MPAs (and hence numbers of sites) and the varying length of time and frequency with which the different MPAs have been surveyed the subsequent analyses to address objective 3 looked at each MPA independently.

Objective 3 - Pt Hicks MNP and Cape Howe MNP

For MPAs with little to no before declaration data but multiple sampling sites, such as Pt Hicks and Cape Howe MNP, the aim was to compare temporal trends in abundance of urchins, abalone and algae at sites within each MPA and its reference sites (Keough *et al.* 2007, Keough and Carnell 2009). In particular, we were interested in whether these sites converge or diverge through time since declaration. Such patterns might be simple linear relationships, or could be more complex functions, incorporating seasonal effects and long-term temporal cycles. Characterising such trends requires a reasonable time series, and confidence in the analysis comes from increasing numbers of sites and lengthening time series.

Monitoring data from Pt Hicks consisted of four sites in the MPA and four reference sites outside (Table 1). These sites have been surveyed four times after declaration of the MPAs. The dataset for Cape Howe MNP, however, was much more complicated. One survey was conducted prior to declaration of the MPA and four after-declaration surveys occurred. However, two of the reference sites which were in *Centrostephanus* barrens were changed for the third after-declaration survey to sites that had lower abundances of *Centrostephanus*. Thus, these latest two surveys were biased for the purpose of the current analysis as they were specifically chosen as sites with low numbers of *Centrostephanus*. One of the MPA sites was also changed due to non-independence. The results we present are for the one before declaration and two after declaration surveys of Cape Howe MNP, with three replicate sites in the MPA and three replicate sites from reference locations.

A repeated-measures analysis of variance (ANOVA) was run for each of the above MPAs. This tested for an effect of MPA overall, an effect of time, or if patterns of abundance of urchins, abalone and algae with regard to MPAs changed over time (i.e. an MPA*Time interaction). If there was an overall effect of MPA, this would suggest that MPA and reference sites were different from the beginning and have remained different. This is in contrast to an interaction, which would indicate a convergence or divergence between MPA and reference sites since declaration. For example, if sites both inside and outside the MPA had high densities of *Centrostephanus* when the MPA was declared, a sign of greater 'resilience' would be a decrease in *Centrostephanus* numbers (and increasing canopy-forming algal cover) inside the MPA while the reference sites stayed the same. While that example represents divergence, if sites both inside and outside an MPA started off with *Centrostephanus* absent or in low abundances, an increase of *Centrostephanus* only in reference sites would indicate 'resistance'.

Objective 3- Beware Reef MS

There is one site inside Beware Reef MS and one reference site, which have been surveyed five times since 2002 (Table 1). For small MPAs without before-declaration data, where there is no

possibility for spatially independent replicate sites, the recommendation is to treat the MPA and reference data sets as two matched time series (Keough *et al.* 2007, Keough and Carnell 2009). Whether the time series are convergent or divergent is tested using a Least Squares Linear Regression analysis. Confidence in this data analysis comes primarily from longer time series. In this analysis, the replicate is each survey, such that the data used in the regression analysis is the number of animals per transect/site (or average algal percent cover).

The most direct way to address questions of differences between MPA and reference sites is to subtract one value from the other. In this analysis, we subtracted the reference site value from the MPA site value. This means that if the value is positive, then there is a greater abundance or cover of the species of interest at the MPA site, while a negative value corresponds with a greater abundance or cover at the reference site. Each replicate observation (survey) then helps us to detect if there are consistent patterns emerging over time. Such patterns may reflect convergence, whereby sites that were initially different have now become more similar (values approaching 0), or divergence, in which sites that were initially similar have now become increasingly different (values either positive or negative). We might also see no relative change, where the slope of the regression line is close to 0. If this is the case and values have simply remained positive over the survey period, this would suggest that initial differences between the two sites have remained over time.

To perform a regression analysis, we also needed to assign a relevant number as the x-value. In regression analysis, the x-value is the 'predictor' or 'independent' variable. In order to choose a meaningful value for each survey, for each survey date we calculated the number of years (to three decimal places) since the declaration of the MPA system in Victoria (16 November 2002). The hypothesis we are then testing is that 'increased time since declaration of an MPA results in a change in the difference between the MPA and reference site'.

Data was also collected for Beware Reef as part of the Reef Life Survey, which uses similar methodology as the SRMP (Barrett and Buxton 2002). However, while there were a number of sites that were surveyed since 2008, only one site within the MPA has been surveyed consistently and no site has been surveyed across multiple years outside the MPA (Table 2). The large number of surveys gives us a better indication of temporal variation inside the MPA than the SRMP, but makes an assessment for the current purpose difficult as we have no clear indication of patterns outside. This means we are unsure of whether patterns inside the MPA match those outside or are showing a unique pattern. Possibilities for displaying the data inside the MPA were explored (e.g. displaying just the 'Auckland' site or averaging across the multiple MPA sites when conducted in the same season), however as this does not aid in the current assessment are not presented here. Consequently, no data analyses were conducted on the Reef Life Survey dataset.

Objective 3- Wilsons Promontory MNP

For MPAs with before- and after-declaration data and multiple sites, the aim is to estimate changes before and after declaration at each site, and to compare before-after changes occurring at sites inside and outside of MPAs (Keough *et al.* 2007, Keough and Carnell 2009). Data consist of time series for each site, but the aspect of interest is the average condition of each site before and after declaration and how this average condition has changed between before and after. At Wilsons Promontory we used 20 out of 28 sites: those that had been sampled consistently (i.e. in at least 10 of the 12 surveys). This included sites inside and outside of the MPA, and on the East and West coasts of Wilsons Promontory. Each coast included equal numbers of sites inside and outside the MNP (Table 1).

The basic statistical model for this MPA is the MBACI design of Keough & Mapstone (1995) and Downes *et al.* (2002), which was used by Edgar and Barrett (1997; 1999), and which compares the

change in the abundance and diversity of species (or values of any other variable) at sites at a time before and after the declaration of the MPA. It also compares whether sites outside the MPA change differently from those inside (see also Keough *et al.* 2007 and Power and Boxshall 2007). The replication for this analysis is based on the numbers of sites. Data in this case were analysed using a two factor ANOVA, with the factors being Status (MPA vs Reference) and Coast (East vs West). Status and Coast were considered fixed factors and were crossed within the analysis. The data used in this analysis was the change in abundance of *Centrostephanus*, abalone, wrasse and algae at each site between surveys conducted before and after establishment of the MPA (see below).

The MBACI test functionally uses sites as the unit of replication, and compares changes Before and After at each site. Multiple surveys are used to provide reliable estimates of the state of a particular site for a particular time period, but are not directly part of the statistical calculations. To simplify the computation and the presentation of results, we calculated Before and After means for each site, and then the Before-After change for that site. We then compared the B-A changes occurring inside and outside the MPA (see Keough and Carnell 2009 for a more detailed explanation). This test is mathematically identical to the expanded test described by Keough and Mapstone (1995).

Inclusion of Coast was intended to increase power of the sampling by removing large scale spatial variation. We also included in initial analyses two covariate factors recommended by Edmunds *et al.* (2012). These were Exposure Ranking (1-9, where 1 is maximum exposure and 9 is sheltered bays) and Complexity Index (CI) (1-4, where 1 is flat rock reef and 4 is highly structured and high relief reef). Inclusion of Exposure Ranking and Complexity Index was intended to increase power of the sampling by removing large scale spatial variation. If such variation did not exist, power would instead be reduced, by loss of degrees of freedom associated with Sites. We therefore conducted analyses involving these two co-variables, but if they had little effect, we omitted it from the final statistical model. We used a conservative α of 0.30 for this step.

Objective 3 – species of interest

While *Centrostephanus* numbers were of most interest, the abundance of a few other key reef species was also considered. This included the ‘habitat-forming algae’, a functional grouping of the brown canopy-forming Laminariales and Furoids: *Ecklonia radiata*, *Phyllospora comosa*, *Durvillea potatorum*, *Sargassum* spp., *Cystophora* spp., *Calyocystis uvifera* and *Acrocarpia* spp. As other important temperate reef herbivores that could potentially be influenced by *Centrostephanus*, the purple sea urchin *Heliocidaris erythrogramma* and the black-lip abalone *Haliotis rubra* were also analysed. As an important predator of *Centrostephanus* in Tasmania (Ling *et al.* 2009, Ling and Johnson 2012), rock lobsters (hereafter *Jasus* spp.) were considered. *Notolabrus tetricus* (the blue-throat wrasse) a common reef species was also included as a representative fish for which the available monitoring data provided high power to detect changes (Keough and Carnell 2009). At Cape Howe MNP and Beware Reef MS little to no *Jasus* spp. were recorded on transects, and so were not analysed or graphed.

Insufficient size data for *Jasus* meant we were unable to conduct size analysis on this species. Instead, size data collected for *Haliotis rubra* and *Notolabrus tetricus* were explored. Two metrics were calculated for *H. rubra*: the mean size per survey (across all the abalone measured) and the proportion measured that were above the current legal size for collection in Victoria (110 mm). For *N. tetricus* the total biomass was estimated as well as the biomass above the legal size limit (280 mm). As the size categories used to estimate fish size were grouped into 50 mm categories, the biomass above 300 mm was calculated. This was done by taking the mid-value of each size category (250-300 mm = 275 mm) and multiplying this by a length-weight relationship developed for fish in Victoria (Smith *et al.* 2003). The data for abalone size provided by Parks Victoria was incomplete for Cape Howe MNP and so was not analysed.

Results

Objective 1

As described in the methodology, we examined the dataset against a number of properties that are required to conduct the statistical assessment. While it fulfilled these, how well the SRMP currently informs us of 'resilience' can only be assessed from the outcomes of the analysis itself.

Objective 2

Centrostephanus was first observed in the four most eastern Victorian MPAs at different times (Figure 2). *Centrostephanus* appeared in Pt Hicks MPA between 2001 and 2004, in Cape Howe before 2001 (a museum specimen was recorded from Gabo Island in 1973) and in Beware Reef sometime before 2004. At Wilsons Promontory it was first detected at one site in 2001, but its occurrence at sites has been sporadic since then and it was not detected in the latest survey (in 2011). In relation to the historical expanding range of *Centrostephanus*, we present a graph of long-term sea surface temperatures for south-eastern Australia (Figure 3). This graph shows that since the 1960s/70s sea surface temperatures throughout south-eastern Australia, including eastern Bass Strait, have been above the long-term averages.

The relationship between *Centrostephanus* density and percentage habitat-forming algal cover varied between the different MPAs and was strongest at Cape Howe MNP (Figure 4). Here, at low urchin densities cover of canopy-forming algae varied substantially. However, as *Centrostephanus* densities increased there was a strong negative relationship between *Centrostephanus* and black-lip abalone (*Haliotis rubra*). The relationship was not plotted for Wilsons Promontory due to the low number of *Centrostephanus* counted during surveys. The subsequent sections present analyses for the individual MPAs considered.

Objective 3

Cape Howe MNP

There was substantial variation in *Centrostephanus* numbers between sites inside and outside the MPA and no difference between MPA and reference sites was detected (Table 3, Figure 5). This was driven by high abundances of *Centrostephanus* at two of the three reference sites with on average 1,000 individuals over the transect (or 5 m⁻²). Correspondingly, the canopy-forming brown algal cover was also low, with between 10-20% cover at these sites. However, at one of the reference sites *Centrostephanus* was barely present and canopy cover was high, and the opposite was found at one of the MPA sites with moderate densities of *Centrostephanus* (365 or 1.8 m⁻²) and lower cover of habitat-forming algae. Very few *Jasus* spp. were observed across the entire survey period (four in total; three at reference sites before declaration of the MPA, and one at a reference site after declaration) and none in the latest surveys in 2009 and 2010.

The abundance patterns for the other analysed species over the monitoring period did not appear to change whether the site was in the MPA (Table 3). There was, however, an overall effect of the MPA for *Haliotis rubra* (black-lip abalone), which indicates that these have consistently been in greater numbers in the MPA compared to reference sites (Figure 5). *Notolabrus tetricus* showed significant changes over time across the sampling sites, but this did not vary in relation to the MPA (Figure 5).

There was no effect of MPA or time on the total biomass or biomass of *N. tetricus* over the legal catch size (Figure 6). The data for *H. rubra* size was incomplete and so could not be analysed.

Pt Hicks MNP

There was no difference between the MPA and outside areas over time detected for the species that we analysed (Table 4). There were relatively low numbers of *Centrostephanus* in the MPA and reference sites (Figure 7). The one exception was a reference site with 37-85 over the survey period on the 200 m transect. The lack of consistency across sites means that we cannot be confident this is associated with an MPA effect. Similarly, despite there being on average between 24-47 times the number of the purple sea urchin *Heliocidaris erythrogramma* in reference sites than in the MPA, the absence of this urchin at one of the reference sites at all four surveys meant that between-site variability was large enough that there was no significant MPA effect. Interestingly, despite this trend for greater urchin numbers at reference sites, habitat-forming algae showed no significant change in abundance over time, but across all surveys was in greater abundance at the reference sites compared to sites within the MPA (Figure 6). The number of rock lobster (*Jasus* spp.) appears higher in the MPA, particularly soon after the MPA was declared, but the high variability between sites meant that no significant difference was detected.

The black-lip abalone (*H. rubra*) showed an increase in abundance over time, and while the trend was for increasing numbers in the MPA, there was no significant difference between the MPA and sites outside (Figure 6). However, it is possible that a significant difference would be detected if this trend was to continue (the last survey was 2010), with the more powerful statistical test that would results from a longer time series

The proportion of black-lipped abalone (*H. rubra*) greater than the legal size limit (110 mm) was higher in the MPA than the reference site across all surveys (Table 4, Figure 8). The average abalone size at MPA sites was 138 mm compared to 120 mm in the reference sites or a 14% difference. It is difficult to determine if this represents site differences or a rapid effect of the MPA as there was no before data collected, but the difference between sites appears to be stable over time.

There was no effect of MPA on total biomass or biomass greater than 30 cm of *N. tetricus* (Table 4). The main effect was a difference over time, with a decrease of both across all sites since the first survey (Figure 8).

Beware Reef MS

The outputs of the regression analyses are displayed in Table 5. Here, there are a number of values to note. The regression coefficient represents the slope of a line fitted to the data (either positive, i.e. increasing over time relative to the MPA, or negative, i.e. decreasing relative to the MPA). The p value for this analysis indicates how well the values fit that trend (<0.05 indicates a significant trend). The R² value also represents how well the x value (years since declaration) predicts the y value (species abundance).

There were no significant trends detected in the regression analysis for the example species chosen. The regression equation was a poor fit to the data for most species, as shown by the low R² values. The two exceptions to this were *Ecklonia radiata* and *N. tetricus* (blue-throat wrasse) legal biomass. The trend for the legal biomass of *N. tetricus* was increasing in the MPA relative to the reference site since declaration of the MPA (Figure 9). Conversely, the trend for *E. radiata* was decreasing biomass in the MPA over time relative to the reference site (Figure 8). For linear regression, five samples is a relatively small sample size. The high R² value for these two species suggests a good-fit of the data. The non-significance of the linear regression analysis for these two species could be a

result of this small sample size and these patterns may become clearer with more surveys (Quinn and Keough 2002).

While there was no relationship with respect to time since declaration of the MPA, the number of *Centrostephanus* was consistently greater in the MPA compared to the reference site (Figure 9). At the MPA, the average density varied between surveys from 0.29 m⁻² to 1.35 m⁻². On average, this represented 100 more individuals per transect than at the reference site, which varied between 0.085 m⁻² and 0.235 m⁻².

Similarly, the number, mean size and the proportion of legal size of *H. rubra* (black-lip abalone), were greater in all surveys in the MPA compared to the reference site. There were on average 37 more abalone per transect, with the abalone on average 13 mm larger, with more than 84% of the population above legal size compared to 55% at the reference site. As with Pt Hicks, it is difficult to determine if this represents site differences or a rapid effect of the MPA as there was no before data collected, but the difference between sites appears to be stable over time.

Wilsons Promontory MNP

Jasus edwardsii (southern rock lobster) was the only species to show a significant change in abundance after declaration of the MNP, slightly increasing in numbers at the MNP sites and decreasing at reference sites (Figure 11). The absolute numbers of *J. edwardsii* were low (<1 per transect), but increased from a before-declaration average of 0.3 per transect to 0.4 after declaration in MPA sites and declining to 0.18 in reference sites. *J. edwardsii* also showed a significant relationship to the exposure index (Figure 13), with decreasing numbers at high exposures (Table 6). Abundance of *Centrostephanus* was low at all sites with a total of 18 individuals counted over the entire survey period.

The predominant effect after declaration was a greater decrease in abundance of several species dependent on coast at both MPA and reference sites (Table 6). For *Heliocidaris erythrogramma* this was a greater decrease on the east coast compared to the west. On average, this represents 26 fewer individuals per transect on the west coast since 2002, and 75 less on the east coast; in other words, almost three times the decrease. For both *H. rubra* and *N. tetricus*, the decrease was greater on the west coast than the east (Figure 11). For *H. rubra* this was 25 less individuals. *H. rubra* also demonstrated a significant relationship with CI Index, where greater decreases since declaration of the MNP occurred at high CI indices (Figure 11).

Discussion

Objective 1

The SRMP methodology is capable of detecting changes in Victoria's sub-tidal reef ecosystems. The SRMP provides a valuable dataset for which trends in a range of species can be assessed. It is worthwhile addressing the fact that there are some things the SRMP does well and some things that it can't do so well. From previous reports, the SRMP generally has better power to detect changes in species or species groups that are relatively common (present in most sites and surveys) (Keough *et al.* 2007, Keough and Carnell 2009). It is more difficult to detect changes in species that are uncommon (eg. *Jasus* spp.) or highly variable between surveys (eg. most fish species). The ability of the data to detect changes related to protection, varies between MPAs based on their size (and therefore replication) and the number of surveys for each MPA. MPAs with either a large number of sites and/or the longest time series provide the best data for assessment. Therefore, for the purposes of the current assessment the ability of the SRMP to assess resistance or resilience is dependent on the species in question (how well the survey design best approximates its abundance) and the particular MPA (spatial and temporal replication).

Currently, Wilsons Promontory MNP has the most extensive dataset of the four MPAs assessed, with a large number of sites and surveys both before and after the MPAs were declared. Cape Howe and Point Hicks MNPs are a similar size, have similar datasets and with increased surveys should be able to provide a stronger assessment of change. However, the switching of sites in the latest surveys at Cape Howe MNP has made the current assessment and interpretation of results very difficult. Finally, the small size of Beware Reef MS means that our ability to detect changes relies on the number of surveys (currently only five). One option to increase the number of surveys at this MPA would be for Reef Life Survey to conduct surveys at the two SRMP sites at least yearly. However, the spatial and temporal replication is only one side of the story. By design the SRMP is meant to estimate the abundance of as many species as possible, but this is not necessarily the best design for the species in question for the current assessment.

For the current report, testing whether MPAs show higher resilience than areas outside to disturbance from *Centrostephanus* depends on the degree of spatial replication and requires a long enough time series to statistically demonstrate that any initial similarities or differences between these areas have changed over time. This is difficult with only a handful of surveys, as the confidence of identifying changes and attributing them to establishment of the MPA (as opposed to yearly and seasonal variability) becomes stronger as the time-series increases. Noting that the evidence from such monitoring programs will always only be correlative, and the focus is on the strength of the correlative evidence (Downes *et al.* 2002). However, the resistance of MPAs is more easily assessed, particularly where the size of MPAs allows for greater spatial replication. This is due to the fact that this only requires a test of whether the presence or abundance of a threat (eg. *Centrostephanus*) is the same or greater outside the MPA.

The range extension of *Centrostephanus* is just one example of the result of climate change having impacts on marine ecosystems, in which there is a clear pathway for MPAs to potentially provide greater resistance or resilience. Currently, we have little understanding of whether MPAs may show similar resistance or resilience to other climate change impacts such as increasing frequency and intensity of extreme weather events, ocean acidification and increasing ocean temperatures. However, habitat-forming algae, such as the ones assessed here, will be important components to monitor over time to assess such questions as many of them already seem to be in decline in some places, or are predicted to decline under warming and disturbance scenarios (Byrnes *et al.* 2011, Smale and Wernberg 2013, Wernberg *et al.* 2013). These species are important as ecosystem engineers that provide

habitat and food for a range of species. Fortunately, habitat-forming algae are relatively well surveyed by the current design. There are also a number of species in other habitats that are expected to be affected by climate stressors (Klemke & Arundel 2013).

Objective 2

Abundances of *Centrostephanus* are highest in the area of Cape Howe MNP, with decreasing densities heading westwards. Sea-surface temperature data indicates a rise above the long-term (recorded since 1880s) temperature average beginning from the 1960s and coincides with reports of *Centrostephanus* at Gabo Island (near Cape Howe MNP) in 1973, the Kent group (Bass Strait) in 1974 and St Helens, Tasmania in 1978. The commercial catch of *Centrostephanus* in eastern Victoria (Fisheries Victoria region 24, between Cape Howe MNP and 19 km east of Pt Hicks), increased from a low of 2 tonnes in 1996/1997 to 17 tonnes in 1997/98 and then remained stable around 20 tonnes for a number of years before increasing to 33 tonnes in 2002/03. In 2003/04, catch dropped back down to 20 tonnes (Hurst 2010). It is unclear if Cape Howe represents a range expansion or if *Centrostephanus* has been present in this area at similar abundances for considerable time. However, *Centrostephanus* was not detected at Pt Hicks in 2001, but was counted both there and at Beware Reef in 2004. This is consistent with other evidence that *Centrostephanus* is a range-expanding species and is increasing its population size due to increasing ocean temperatures and strengthening of the East Australian Current in the area (Ling *et al.* 2008, Ling *et al.* 2009, Johnson *et al.* 2011).

While *Centrostephanus* is present at the four MPAs, it is only at Cape Howe that densities were high enough to show a dramatic influence on habitat-forming algal cover (indicated by a negative correlation between urchin densities and algal cover). Consistent with the relationships at Cape Howe, Ling *et al.* (2009) showed using a similar but larger scale correlative approach that significant impacts on macro-algal percentage cover in Tasmania are observed when *Centrostephanus* densities are $\geq 4 \text{ m}^{-2}$. In NSW, Hill *et al.* (2003) tested the impact of different densities of *Centrostephanus* (0, 2, 4, 10 m^{-2}) in a manipulative experiment and it was only at 10 m^{-2} that an urchin barren occurred. Given the long history of *Centrostephanus* at Cape Howe (described above), some sites were already heavily impacted (urchin barrens and reduced abalone abundances) before the MPA was declared (February 2001 Cape Howe survey). However, at Pt Hicks, Beware Reef and Wilsons Promontory, densities of *Centrostephanus* on transects currently remain insufficient to cause a statistically measurable effect using the available SRMP data. There are two possible reasons for this: 1) this method was not designed specifically for assessing *Centrostephanus* abundances, and the data collected using the current design have substantial variability between sites (denoted by the error bars on the graphs). There is the potential for small barren areas to form that only cover part of the transects and are therefore not picked up using the current survey method. Or 2) this could be linked to dispersal/connectivity mechanisms and the relatively recent arrival of *Centrostephanus* to these areas. While *Centrostephanus* numbers at Pt Hicks seem stable over the survey period, numbers were highest in the latest survey at Beware Reef suggesting an increasing population in this area.

As well as a strong relationship between increasing *Centrostephanus* and declines in habitat-forming algae at Cape Howe, we also documented a strong negative association between increasing *Centrostephanus* densities and a decreasing abalone (*H. rubra*) numbers. This is consistent with previous work in Tasmania (Strain and Johnson 2012) and observations made by abalone divers in the area. It demonstrates the strong influence that *Centrostephanus* may have on this species.

Objective 3

From the analyses conducted, where *Centrostephanus* densities were high enough, they affected *H. rubra* and habitat-forming algae regardless of whether within or outside an MPA. However, irrespective of *Centrostephanus*, we also saw a range of sustained differences in species

abundances inside compared to outside MPAs across the surveys conducted since the MPAs were established. At Cape Howe MNP, this was a greater abundance of abalone at MPA sites and at Pt Hicks there was a sustained difference in average size and proportion of legal sized abalone in the MPA. It is difficult to determine from the SRMP data if this represents pre-existing differences between these MPA and reference sites, or a rapid effect of the MPA. At Wilsons Promontory we documented an overall increased abundance in the MNP and a decrease outside in abundance of lobster after declaration. While we found no statistical differences for the focal species at Beware Reef MS in comparison to its reference site, strong trends were observed for *N. tetricus* legal biomass, with higher biomass above legal size in the MS compared to the reference site. Thus, as we have seen from the current data, while some MPAs currently support a greater abundance and/or size of abalone, future increases in *Centrostephanus* could counteract such differences.

The period in which *Centrostephanus* has been expanding its range coincides with large reductions in the rock lobster fishery in eastern Victoria (Hurst 2010), as well as changes in ocean physico-chemical conditions such as the EAC described above (Klemke and Arundel 2013). In Tasmania, field experiments have demonstrated that large lobsters (greater than 140 mm carapace length) are the main predator of *Centrostephanus* and that in areas in which large lobster were sufficiently dense (2-4 individuals 200 m⁻²) they were able to consume considerable numbers of urchins and maintain habitat-forming algal systems (Ling *et al.* 2009, Ling and Johnson 2012). Given that the primary biological 'defence' of reefs against *Centrostephanus* is the presence of large rock lobster (*Jasus* spp.) in Tasmania, it is possible that the abundance and size of this species may be a critical requirement for resistance to or recovery from the impacts of *Centrostephanus* invasion in Victoria. It is not possible to assess this question using data from the SRMP.

Objective 4 - Dataset limitations and recommendations

The statistical power (or ability) of the SRMP monitoring program to detect meaningful changes has been reviewed previously by Keough *et al.* (2007) and Keough and Carnell (2009) and so was not covered as part of this report. However, one of the potential contributions to a low power to detect change in some species is that the SRMP method aims to estimate the abundance of any algal, invertebrate and fish species encountered. This results in substantial variation within the dataset between sites and over time as a species' relative abundance (common or rare) and potential for temporal (yearly, seasonal, daily) variability will influence how well the SRMP approximates its abundance. If for certain MPAs (eg. those in eastern Victoria) there are specific species of interest (eg. *Centrostephanus* and *Jasus*), then additional, species specific survey techniques could be beneficial for better estimating changes in abundance. For example, in the study by Ling and Johnson (2012), *Jasus* at two MPAs were counted with six replicate belt transects (50 x 4 m) inside and outside, with a standardized search time of 30 minutes per transect. In contrast, multiple 5 x 1 m belt transects are commonly used to survey urchin numbers (Constable 1989, Johnson *et al.* 2005). While the methods of the SRMP provide information on a wide range of species, detecting changes in some species may require additional monitoring effort to yield sufficient statistical power. Thus, if detailed information of the *Centrostephanus* threat is deemed necessary, then additional specific survey methods would need to be developed for these MPAs to appropriately measure both *Centrostephanus* and other species that it may be affecting, such as canopy-forming algae, as well as its lobster predator (*Jasus edwardsii*). However, such monitoring surveys can only provide information about apparent associations among species. Targeted research would be required to understand the interactions between *Centrostephanus* and other species on Victorian reefs, and to compare them to the interactions that have been observed in Tasmania.

Finally, the Reef Life Survey dataset was unable to be used for the current assessment due to lack of consistent monitoring of sites both inside and outside Beware Reef MS (see Methodology for Beware Reef). If the Reef Life Survey program could incorporate monitoring of the sites used in the

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SRMP to provide greater temporal replication, this would dramatically increase the power of analyses using the data for Beware Reef MS.

Conclusions

The SRMP provides data that was useful for the aims of the current project, but had limitations depending on the MPA in question. Given the original aims of the SRMP were different to that of the current analysis, substantial variability in quality of the data between and within MPAs means that power to detect changes was also variable and often poor for the current purpose. There is no evidence from the analyses conducted in this study to suggest greater resistance or resilience to the range expansion of the sea urchin *Centrostephanus rodgersii* in the four Victorian MPAs in which it has been recorded, compared to areas outside the MPAs. While this is the case for the MPAs considered, in Tasmania the protection afforded by MPAs against *Centrostephanus* invasion has been *via* predation by large (>140 mm carapace) rock lobster (*Jasus edwardsii*) on *Centrostephanus*. *Jasus* is currently in low abundance in all of the Victorian MPAs examined. Tasmanian reefs and MPAs may differ from those in Victoria in a number of ways including the top-down (high-order predator driven) or bottom-up (environmentally driven) forcing of ecosystem dynamics. Thus, in Victoria whether changes in management of fisheries (which can influence top-down dynamics) will also influence resistance or resilience to this particular threat is not well understood. The power of the SRMP to detect changes tends to depend on how common a species is and the quantity and quality of data available for each MPA. The SRMP has the ability to test specific hypotheses in relation to differences in the abundance of species inside compared to outside MPAs. However its ability to detect impacts from climate change will vary depending on how well the survey method suits the species in question and its ability to pick up temporal changes in the abundance of that species.

Acknowledgements

Parks Victoria is acknowledged for provision of reef monitoring data. VEAC provided financial support for this analysis. Dr Jo Klemke provided advice on clarifying project objectives. Krystina Mossop is thanked for editing earlier versions of this report.

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Table 1. Summary details of the MPAs considered in this report.

Marine Protected Area	# Sites MNP/Ref	MPA size (ha)	# surveys before MPA declaration	# surveys after MPA declaration
Cape Howe MNP	4/4	4,050	1	4
Pt Hicks MNP	4/4	4,000	0	4
Beware Reef MS	1/1	220	0	5
Wilsons Promontory MNP	11/9	15,550	7	5

Table 2. Summary details of Reef Life Survey sites, number of surveys each year and the cumulative total.

Status	Site Name	Site Code	2008	2009	2010	2011	2012	2013	Survey Total
MPA	Auckland	BR4	7	7	5	4	4	2	29
	Nth Pinnacle	BR5	1	1			1		3
	Mid Pinnacle	BR6	1		3				4
	Southern Flat	BR7			2				2
	Sth Pinnacle	BR8	1	2	1	1			5
	NE Beware Reef	BR11		1	1	2	3		7
	Ridge Park	BR12	3	3	3	1			10
	Seal Gully	BR14				1	3	1	5
Reference	East Yeerung	BR3		1	2				3
	Pearl Point	BR9	1						1
	East Pearl Point	BR10		3					3
	Outer East Yeerung Reef	BR13			1				1
	Cape Conran Bay	VIC13		1					1
	West Yeerung Beach	VIC15				1			1

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Table 3. Summary of analyses for Cape Howe MNP.

Probabilities in bold indicate a significant effect ($p < 0.05$).

Species	MPA	Time	Time*MPA
<i>Notolabrus tetricus</i>	0.206	0.002	0.062
<i>Centrostephanus rodgersii</i>	0.198	0.598	0.421
<i>Haliotis rubra</i>	0.002	0.313	0.330
<i>Heliocidaris erythrogramma</i>	0.536	0.784	0.586
Canopy Forming Browns	0.313	0.078	0.119
<i>N. tetricus</i> total biomass	0.647	0.054	0.591
<i>N. tetricus</i> legal biomass	0.957	0.083	0.654

Table 4. Summary of analyses for Pt Hicks MNP

Probabilities in bold indicate a significant effect ($p < 0.05$).

Species	MPA	Time	Time*MPA
<i>Notolabrus tetricus</i>	0.894	0.093	0.636
<i>Centrostephanus rodgersii</i>	0.313	0.367	0.442
<i>Haliotis rubra</i>	0.607	0.011	0.099
<i>Heliocidaris erythrogramma</i>	0.070	0.373	0.725
<i>Jasus verreauxi</i>	0.276	0.421	0.446
Canopy Forming Browns	0.025	0.077	0.833
Encrusting corallines	0.843	0.117	0.860
<i>N. tetricus</i> total biomass	0.866	0.000	0.355
<i>N. tetricus</i> legal biomass	0.705	0.002	0.320
<i>H. rubra</i> mean size	0.000	0.500	0.310
<i>H. rubra</i> proportion legal	0.001	0.499	0.096

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Table 5. Summary of regression analyses for Beware Reef MS

Probabilities in bold indicate a significant effect ($p < 0.05$).

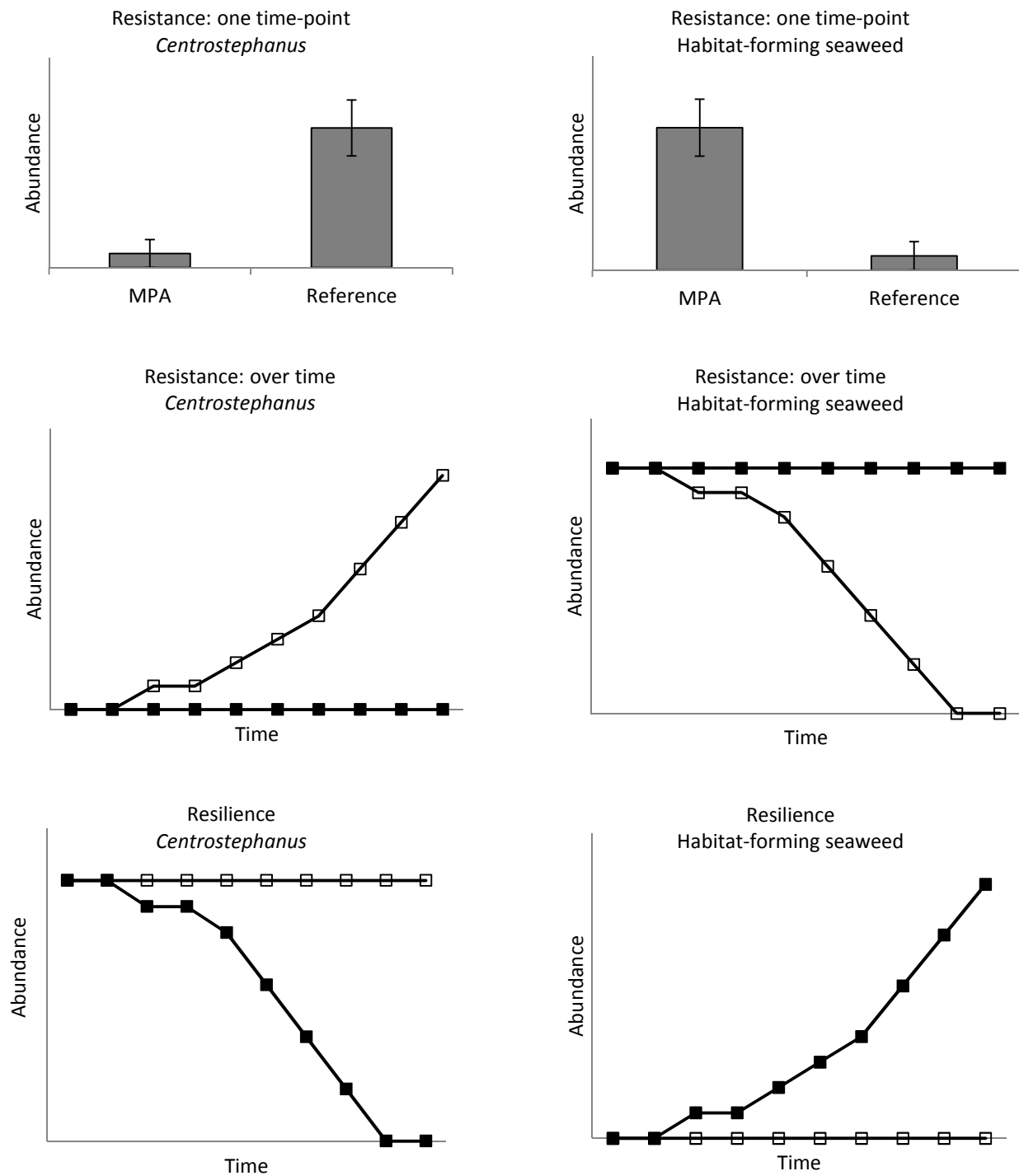
Species	MPA	Regression coefficient	R ²
<i>Notolabrus tetricus</i>	0.414	-1.341	0.230
<i>Centrostephanus rodgersii</i>	0.673	6.423	0.067
<i>Haliotis rubra</i>	0.509	4.018	0.157
<i>Heliocidaris erythrogramma</i>	0.366	4.542	0.273
Canopy forming browns	0.378	-2.708	0.262
<i>Ecklonia radiata</i>	0.052	-2.674	0.767
Encrusting corallines	0.664	-0.638	0.072
N. tetricus total biomass	0.861	-0.032	0.012
N. tetricus legal biomass	0.118	0.366	0.611
<i>H. rubra</i> mean size	0.639	0.205	0.083
<i>H. rubra</i> proportion legal	0.837	0.226	0.017

Table 6. Summary of analyses for Wilsons Promontory MNP

Probabilities in bold indicate a significant effect ($p < 0.05$). Exp is the Exposure ranking and CI is the Complexity Index from Edmunds *et al.* (2012).

Species	MPA	Coast	MPA*Coast	Exp	CI
<i>Notolabrus tetricus</i>	0.820	0.002	0.182	-	-
<i>Centrostephanus rodgersii</i>	0.199	0.249	0.134	-	-
<i>Haliotis rubra</i>	0.481	0.004	0.947	0.293	0.006
<i>Heliocidaris erythrogramma</i>	0.309	0.010	0.341	-	-
<i>Jasus edwardsii</i>	0.042	0.328	0.236	0.031	0.106
Canopy Forming Browns	0.104	0.176	0.538	-	-
N. tetricus total biomass	0.330	0.139	0.439		
N. tetricus legal biomass	0.177	0.232	0.712		
<i>H. rubra</i> mean size	0.630	0.217	0.775		
<i>H. rubra</i> proportion legal	0.622	0.061	0.642		

Figure 1. Hypothetical patterns demonstrating the ideas of ecological resistance (at either single or multiple time-points) and resilience for *Centrostephanus* (left panel) and habitat-forming seaweeds (right panel) in MPAs (filled squares) and reference sites (open squares).



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Figure 2. a) Current range of *Centrostephanus rodgersii* in south-east Australian waters (Image taken from Google Earth and overlain with range lines from Johnson *et al.* 2010). Green line indicates historic range, yellow line indicates range expansion b) Map of Marine Protected Areas (MPAs) considered in this analysis in which *Centrostephanus* has been recorded. c) Mean (\pm SEM) number of *Centrostephanus* per transect (values were averaged across the after declaration surveys) at MPA and reference sites at each of the four MPAs: WP, Wilsons Promontory MNP, BR; Beware Reef MS, PH; Pt Hicks MNP, CH; Cape Howe MNP. At MPAs with low numbers, numbers above bars represent the mean value.

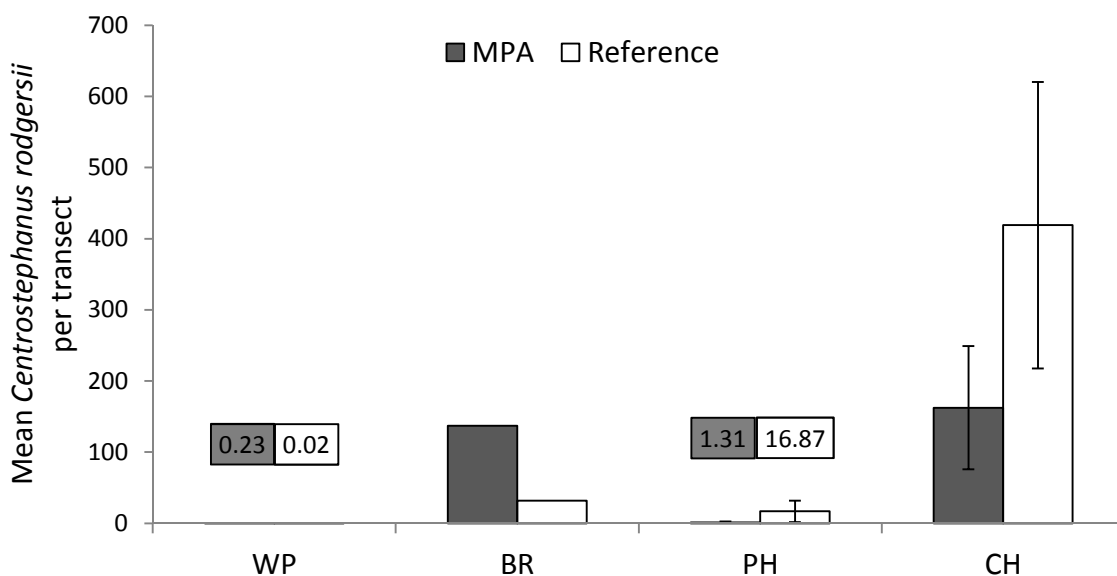
a)



b)

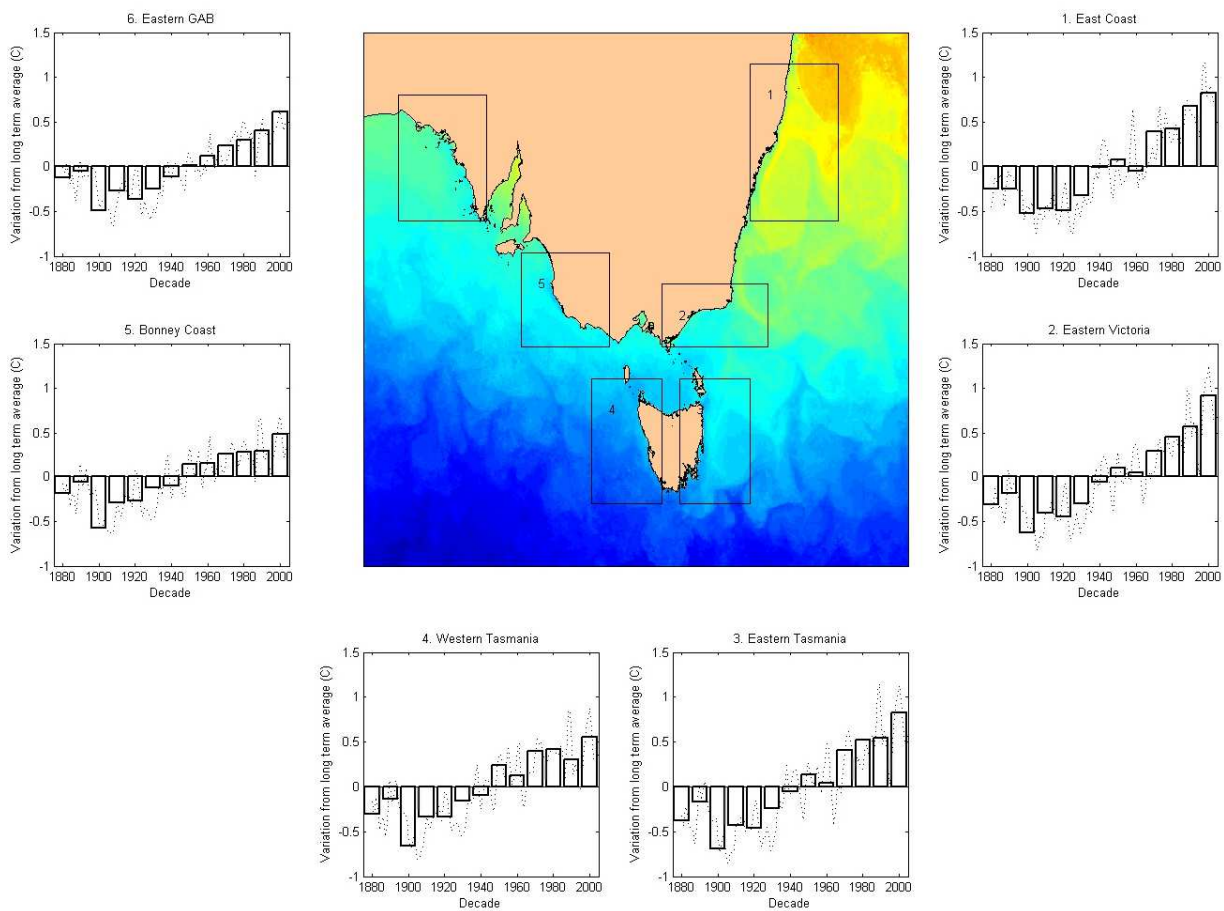


c)



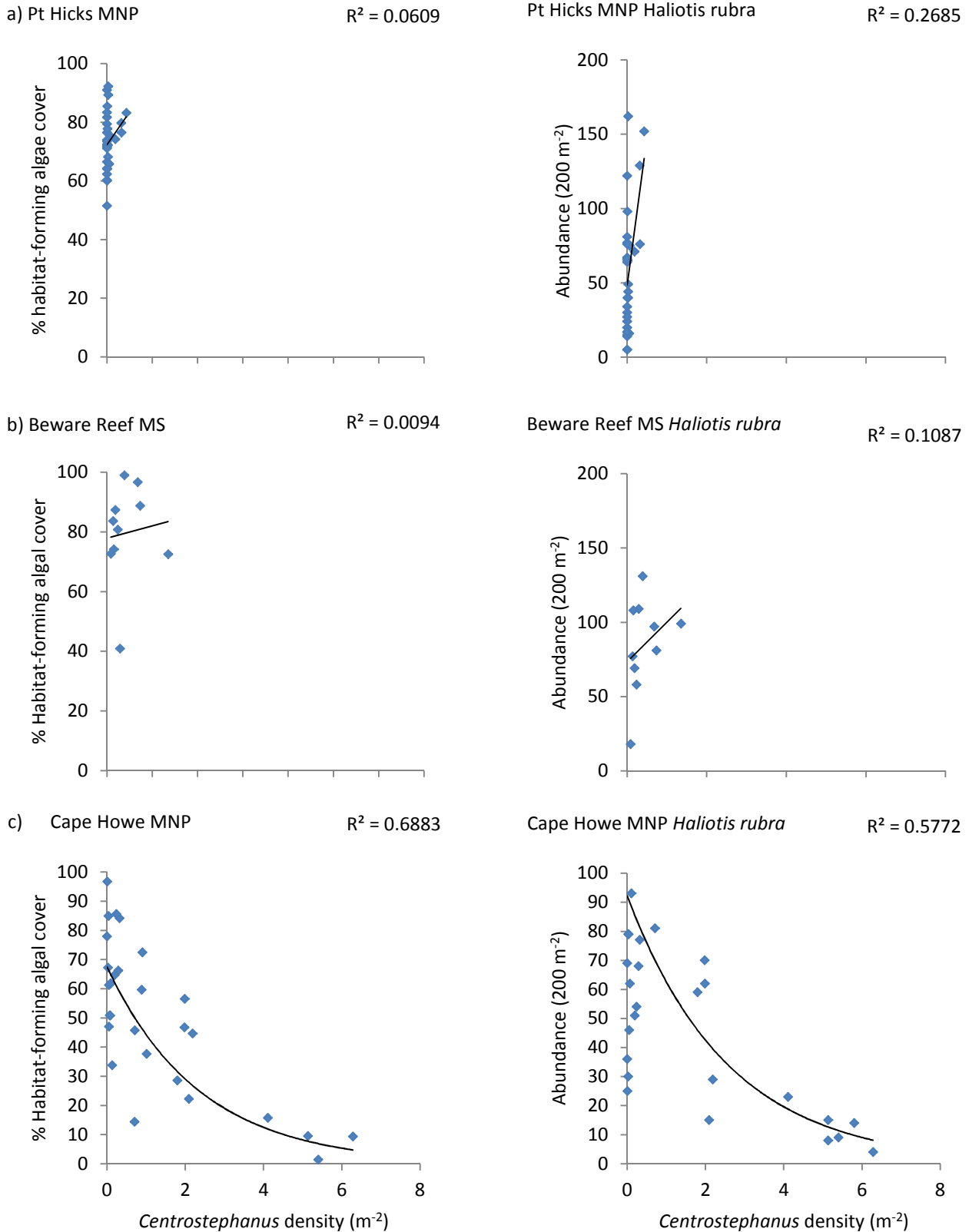
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Figure 3. Sea surface temperature in southern Australia from 1880 to 2000. Note area 2: Eastern Victoria and that in general, water temperatures start to rise above the long-term average after about the 1960s. (Image: CSIRO. [Data was sourced from the Integrated Marine Observing System \(IMOS\) - IMOS is supported by the Australian Government through the National Collaborative Research Infrastructure Strategy and the Super Science Initiative](#))



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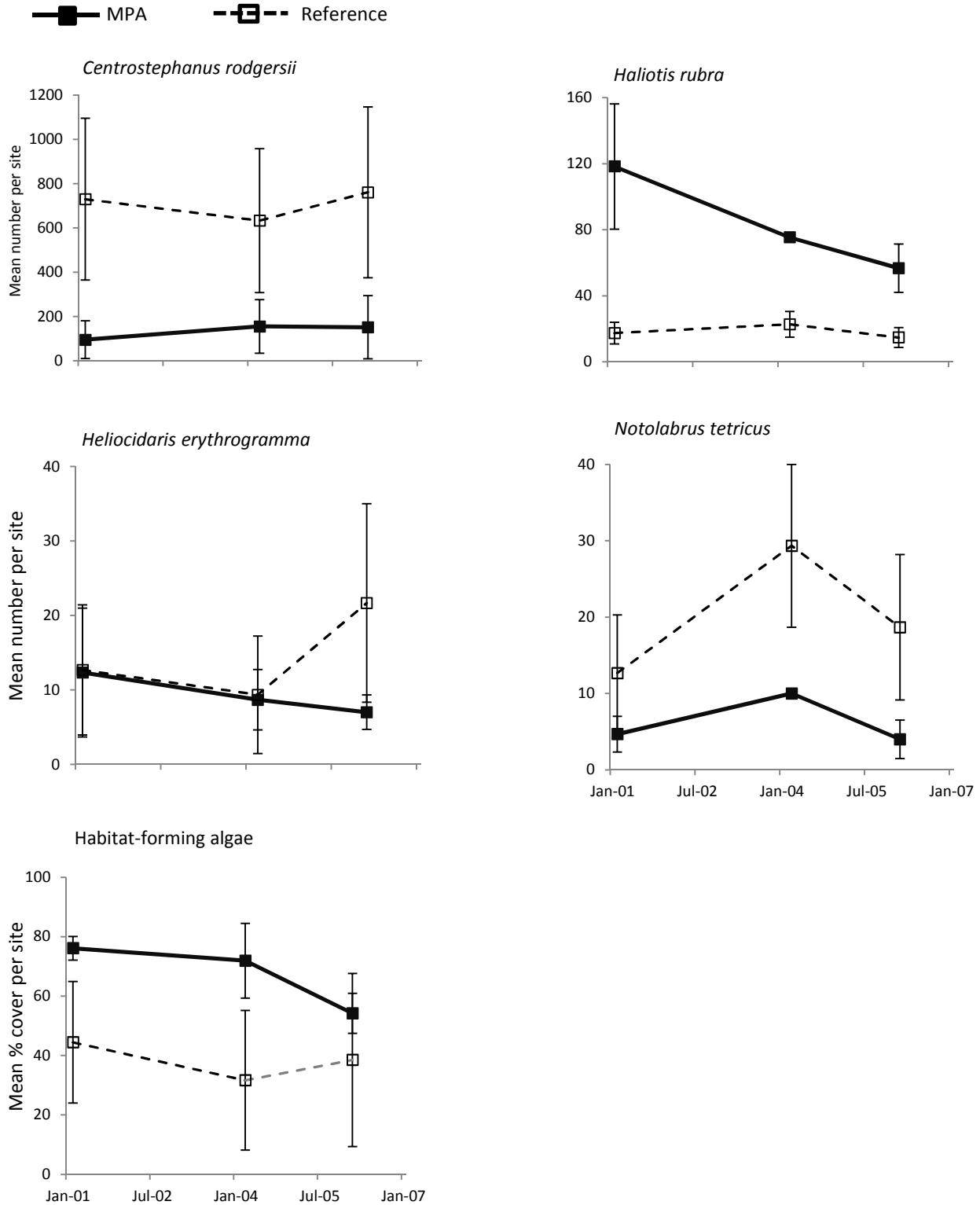
Figure 4. Relationships between *Centrostephanus* density (m^{-2}) and % canopy-forming algal cover at a) Point Hicks MNP, b) Beware Reef MS and c) Cape Howe all surveys included. The line represents the linear trendline fitted to the data and the R^2 value represents the 'fit' of the line.



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Figure 5. Abundances of the focal species at Cape Howe MNP

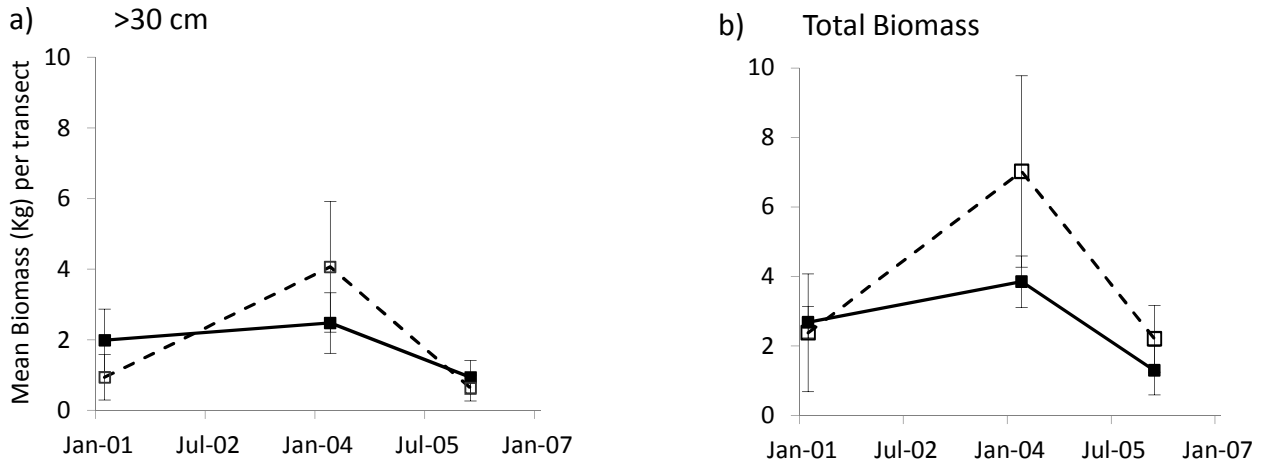
Mean (\pm SEM) abundance of focal invertebrate, algal and fish species at Cape Howe MNP over the survey period before and after declaration of the MPAs, separated into MPA (shaded) and reference sites (clear).



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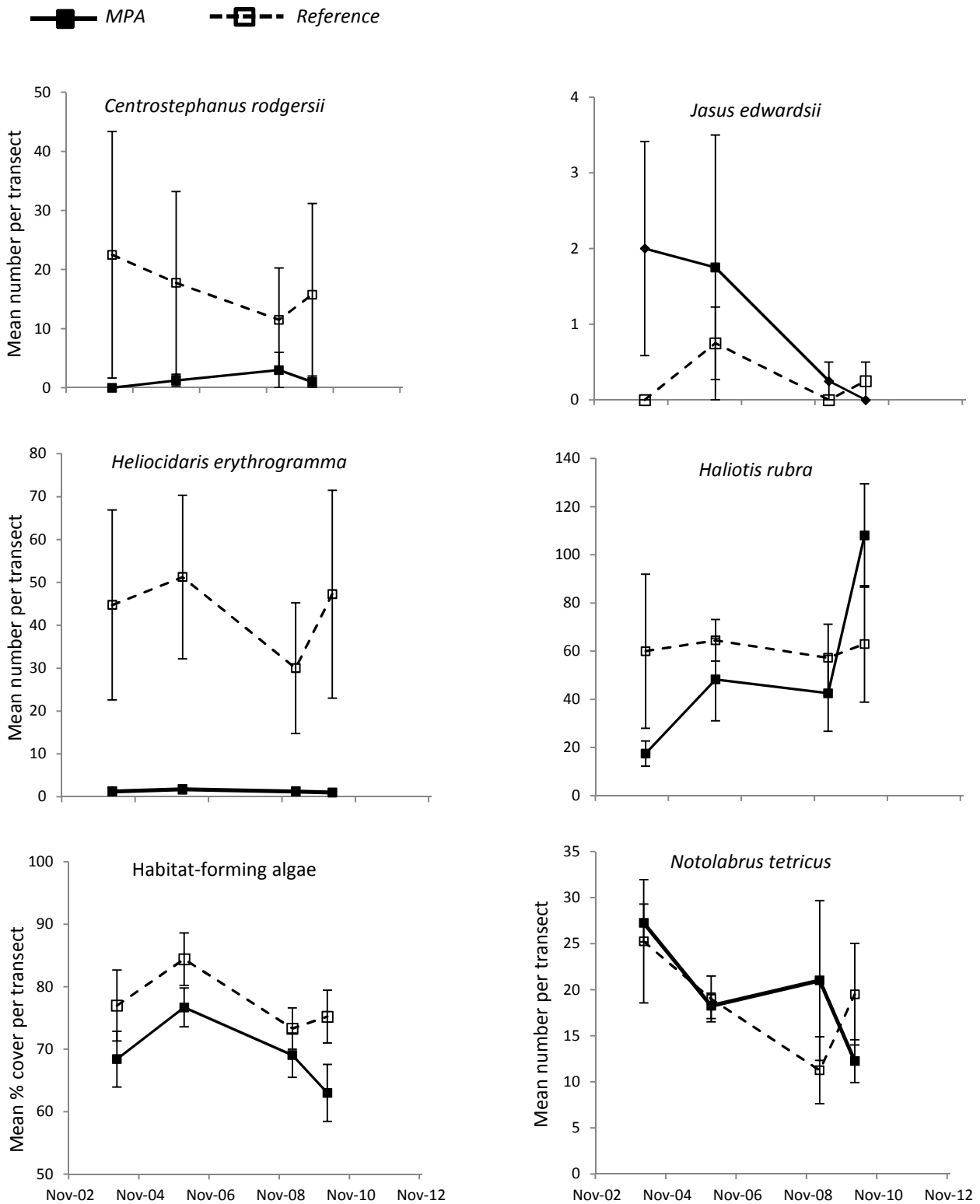
Figure 6. *Notolabrus tetricus* biomass measurements at Cape Howe MNP. Values are the mean (\pm SEM) of a) biomass greater than 30 cm (legal size) and b) total biomass, separated into MPA (shaded) and reference sites (clear).

—■— MPA - - □ - - Reference



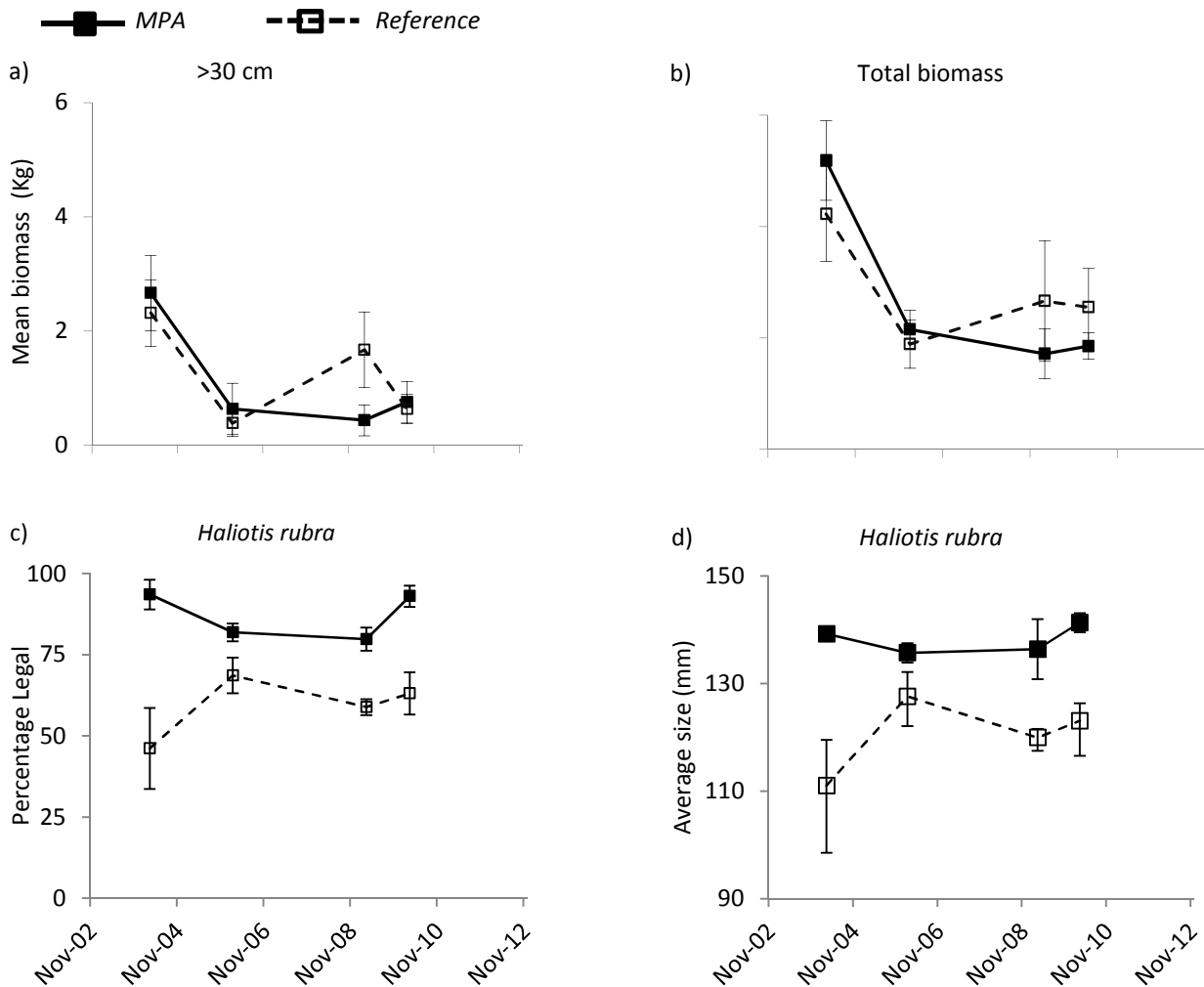
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Figure 7. Mean (\pm SEM) abundance of focal invertebrate, algal and fish species at Pt Hicks MNP over the survey period after declaration of the MPAs, separated into MPA (shaded) and reference sites (clear).



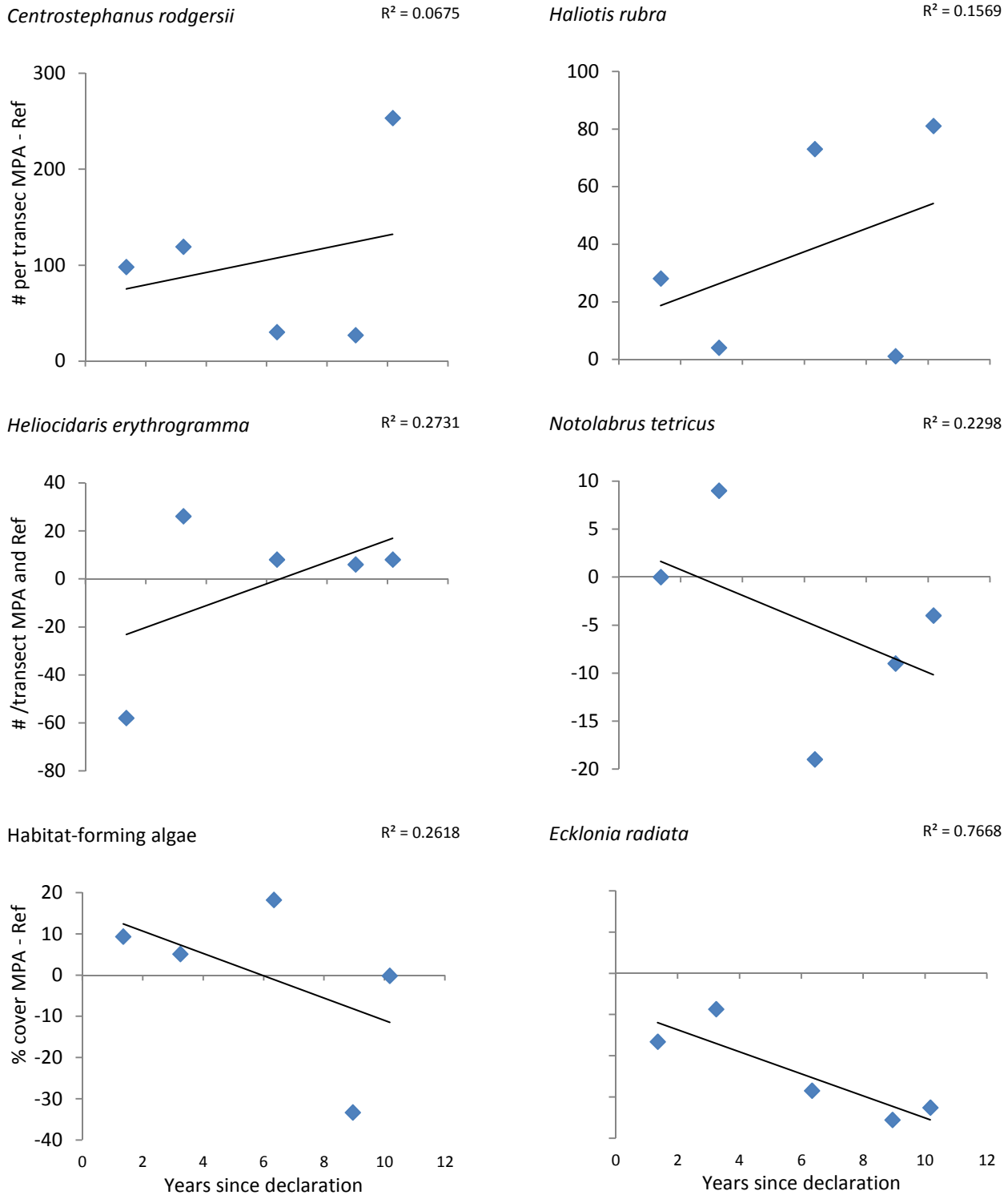
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Figure 8. Mean (\pm SEM) biomass (a, b) of *Notolabrus tetricus* and percentage legal (c) and mean size (d) (\pm SEM) of *Haliotis rubra* at Pt Hicks MNP over the survey period after declaration of the MPAs.



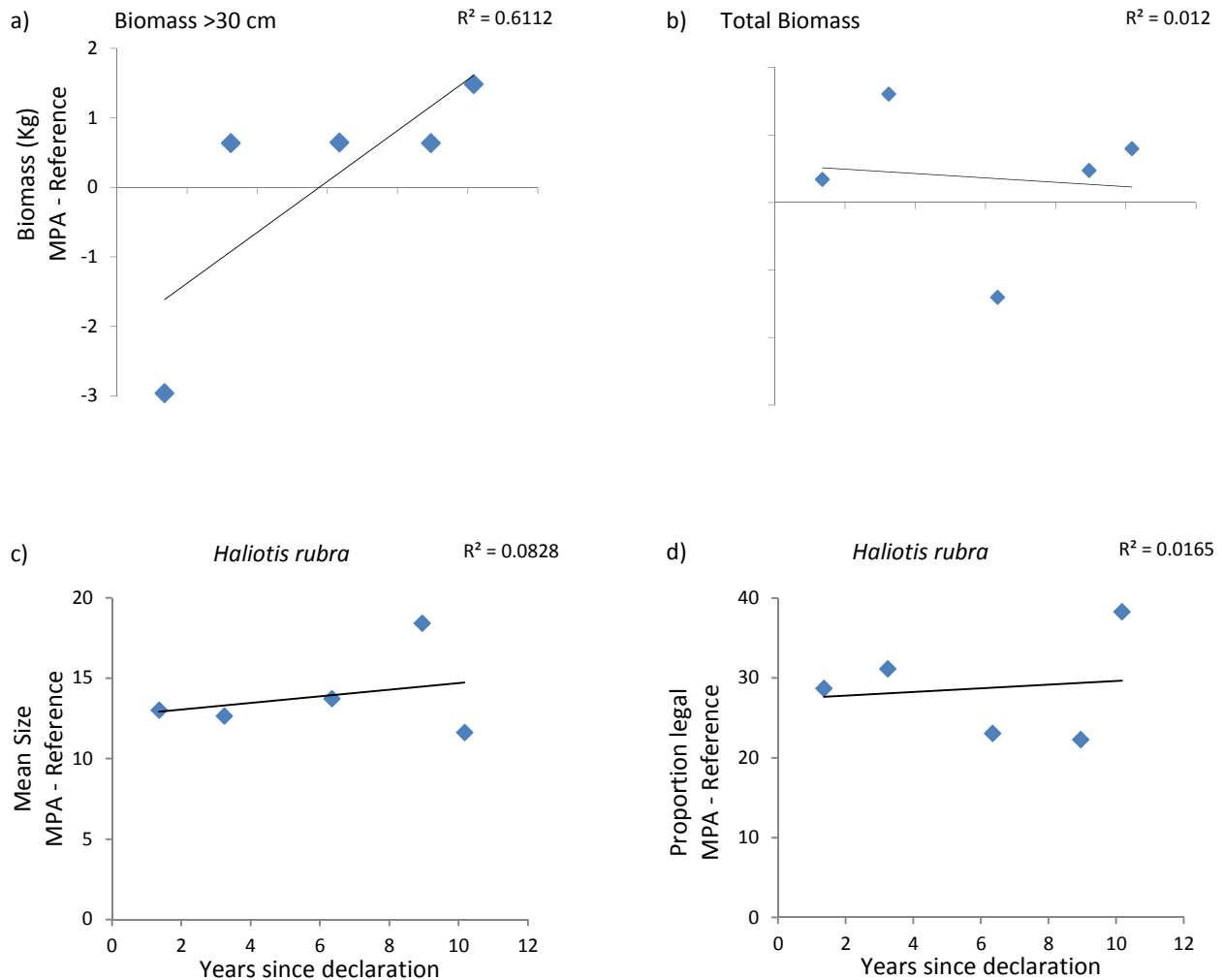
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Figure 9. Focal species at Beware Reef MS. Values are the survey totals for animals and quadrat averages for algae with the MPA value minus the reference site value. Line represents the trend line fitted to the data with the R^2 value presented.



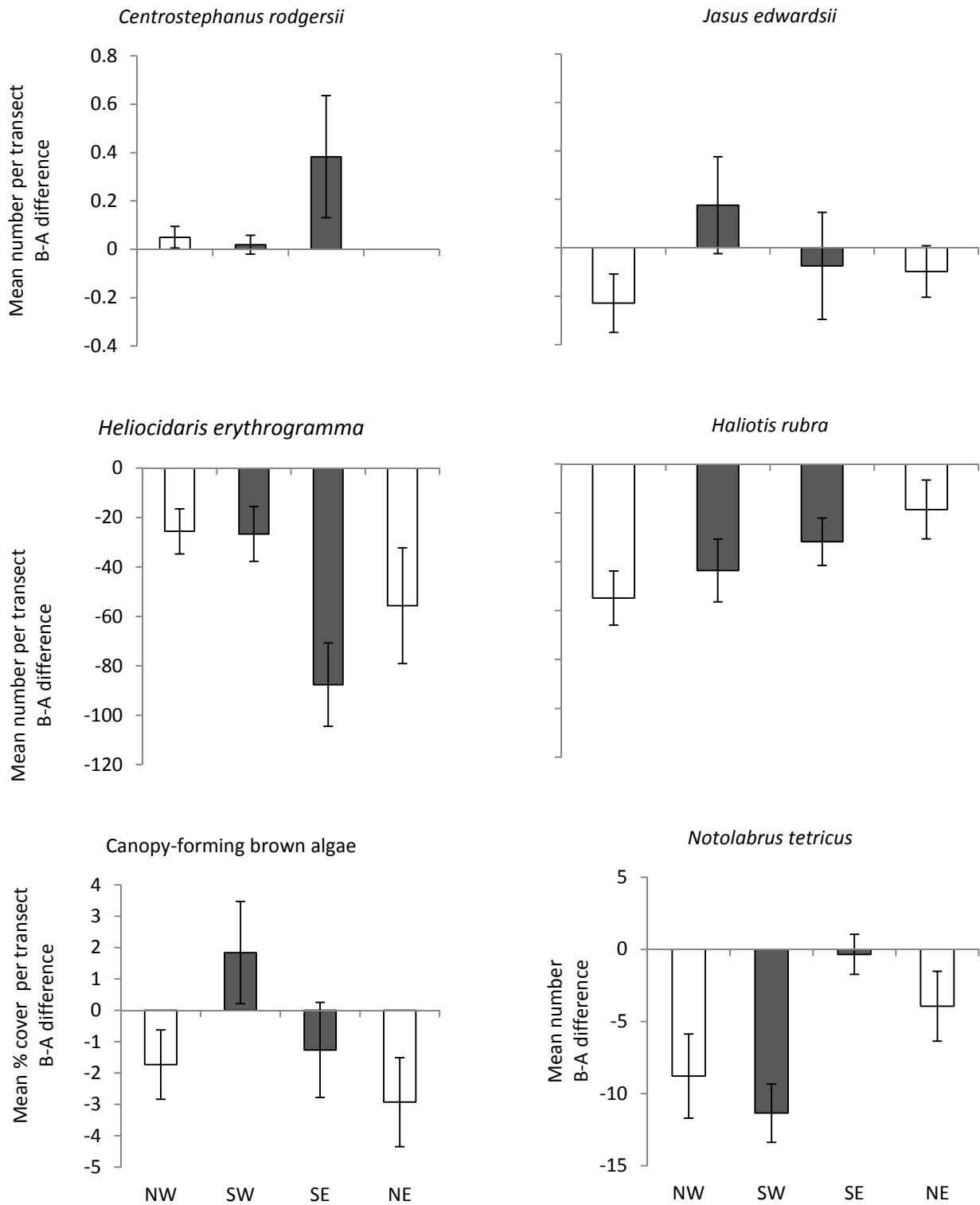
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Figure 10. *Notolabrus tetricus* and *Haliotis rubra* size measurements at Beware Reef MS with the MPA minus the reference site value presented. Graphs for *N. tetricus* are a) Biomass greater than 30 cm (legal size) and b) total biomass. Graphs for *H. rubra* are the c) mean size per survey and d) Proportion above legal size (110 cm) per survey. Line represents the linear trendline fitted to the data with the equation and R^2 value presented.



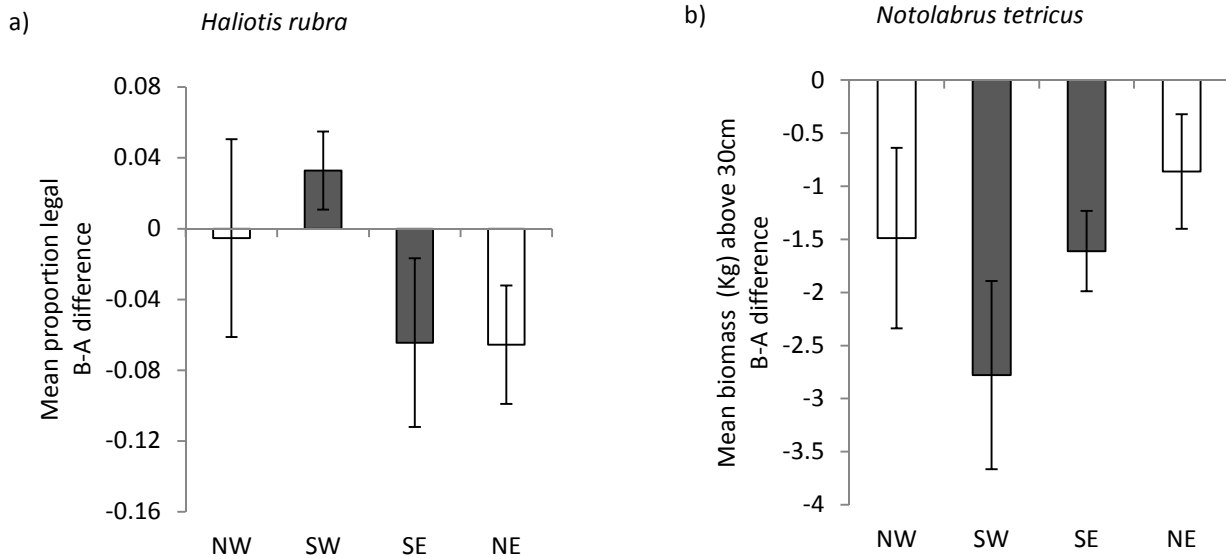
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Figure 11. The mean before–after declaration difference at sites in the various zones within (shaded) and reference (clear) locations at Wilsons Promontory MNP. Reserve/reference zone sites are located in the north-west (NW) and north-east (NE), while MNP sites are located in the south-west (SW) and south-east (SE) of sections of Wilsons Promontory.



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Figure 12. The mean before –after declaration difference of a) *Haliotis rubra* proportion legal size and b) *Notolabrus tetricus* mean biomass above 300 mm at sites in the various zones within (shaded) and reference (clear) locations at Wilson’s Promontory MNP. Reserve/reference zone sites are located in the north-west (NW) and north-east (NE), while MNP sites are located in the south-west (SW) and south-east (SE) of sections of Wilsons Promontory.



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Figure 13. Relationship between mean before-after declaration difference and a) Complexity Index (CI) for *Haliotis rubra* and b) Exposure index for *Jasus edwardsii* at Wilsons Promontory MNP.

