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Anticipating changes to future connectivity within a network of marine protected areas

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Abstract

Continental boundary currents are projected to be altered under future scenarios of climate change. As these currents often influence dispersal and connectivity among populations of many marine organisms, changes to boundary currents may have dramatic implications for population persistence. Networks of marine protected areas (MPAs) often aim to maintain connectivity, but anticipation of the scale and extent of climatic impacts on connectivity are required to achieve this critical conservation goal in a future of climate change. For two key marine species (kelp and sea urchins), we use oceanographic modelling to predict how continental boundary currents are likely to change connectivity among a network of MPAs spanning over 1000 km of coastline off the coast of eastern Australia. Overall change in predicted connectivity among pairs of MPAs within the network did not change significantly over and above temporal variation within climatic scenarios, highlighting the need for future studies to incorporate temporal variation in dispersal to robustly anticipate likely change. However, the intricacies of connectivity between different pairs of MPAs were noteworthy. For kelp, poleward connectivity among pairs of MPAs tended to increase in the future, whereas equatorward connectivity tended to decrease. In contrast, for sea urchins, connectivity among pairs of MPAs generally decreased in both directions. Self-seeding within higher-latitude MPAs tended to increase, and the role of low-latitude MPAs as a sink for urchins changed significantly in contrasting ways. These projected changes have the potential to alter important genetic parameters with implications for adaptation and ecosystem vulnerability to climate change. Considering such changes, in the context of managing and designing MPA networks, may ensure that conservation goals are achieved into the future.

Keywords: *Centrostephanus rodgersii*, climate change, dispersal, *Ecklonia radiata*, kelp, marine reserve, ocean currents, urchin

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Introduction

Continental boundary currents are projected to change under climate change (Wu *et al.*, 2012; Sen Gupta *et al.*, 2015). In particular, western boundary currents (WBCs) globally are forecast to strengthen due to an ongoing poleward shift of the latitude where the maximum wind stress curl occurs (Cai, 2006). Given that continental boundary currents often determine dispersal and large-scale connectivity among populations of marine organisms (Coleman *et al.*, 2011a, 2014; Cetina-Heredia *et al.*, 2015), changes to these currents may have dramatic implications for species distribution, population genetics and population persistence with resultant alterations to entire ecosystems (Verges *et al.*, 2014; Provost *et al.*, 2016).

Networks of marine protected areas (MPAs) are an important tool for safeguarding regions crucial to metapopulation persistence (e.g., source and sink populations), and maintaining levels of connectivity (Palumbi, 2003; Roberts *et al.*, 2003; Coleman *et al.*, 2011b), which may not only ensure persistence but the long-term resilience of populations (Mumby *et al.*, 2011; Magris *et al.*, 2014; Andrello *et al.*, 2015). The size, spacing and arrangement of MPAs relative to scales of dispersal and life history of organisms (Shanks *et al.*, 2003; Durrant *et al.*, 2014), combined with local- and regional-scale oceanography (Roberts, 1997) and other environmental factors, determine the extent to which MPAs are connected and contribute to conservation goals. Moreover, given that both spatial (Castorani *et al.*,

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2015) and temporal (Blowes & Connolly, 2012) variability in dispersal distance and spread affects the likelihood of population persistence, understanding variability in dispersal and connectivity is a key component of MPA network planning (Pendoley *et al.*, 2014). Future changes to connectivity should thus ideally be considered and anticipated in conservation initiatives to allow appropriate review of MPA designs and to ensure that MPAs achieve their conservation goals both now and into the future (Magris *et al.*, 2014; Parsons *et al.*, 2014).

Despite the acknowledged need to consider both extant and future connectivity, as well as its shorter-term temporal variability, in conservation planning, this task is challenging and there is much uncertainty and subjectivity in how to incorporate estimates of connectivity for multiple species and climate scenarios into management initiatives (Mundy *et al.*, 2009; Mumby *et al.*, 2011; Magris *et al.*, 2014; Pendoley *et al.*, 2014; Melia *et al.*, 2016). The handful of studies that quantitatively examine future connectivity in a conservation context reveal significant changes to connectivity and self-seeding among MPAs that is mediated by concurrent change in species reproductive timing, larval survival and development (Andrello *et al.*, 2015). Moreover, current spatial arrangements of MPAs are often not adequate to protect future thermal refugia that may confer long-term resilience and persistence to marine communities (Magris *et al.*, 2014; but see Almpanidou *et al.*, 2016) and might not even be spaced to provide effective current protection (Pendoley *et al.*, 2014). Adaptively managing networks of MPAs that will continue to achieve conservation goals into the future will thus be dependent on understanding future changes to connectivity as well as the ability of species to genetically adapt to climatic change (Mumby *et al.*, 2011).

A predicted 'hotspot' of climatic change is the east coast of Australia which is bathed by the East Australian Current (EAC), a poleward flowing western boundary current that has been steadily increasing in strength (Ridgway, 2007; Hill *et al.*, 2008; Cetina-Heredia *et al.*, 2014), and is predicted to further increase by 12% in core transport and by 35% in poleward extension by 2060 (Sun *et al.*, 2012). Transport in eddies within the EAC has also increased in the last three decades (Cetina-Heredia *et al.*, 2014), and longer-lived anticyclonic eddies are expected to occur more frequently (Matear *et al.*, 2013; Oliver *et al.*, 2015). These changes are likely to impact dispersal and connectivity within the EAC (Cetina-Heredia *et al.*, 2015), with implications for critical population parameters (e.g., genetic diversity, inbreeding), as well as species ranges (Ling *et al.*, 2008) and ecological interactions (Johnson *et al.*, 2011; Verges *et al.*, 2014).

Along the east coast of Australia, there is a network of temperate MPAs spanning over 1000 km of shoreline (Coleman *et al.*, 2015; Fig. 1). Extant genetic connectivity within and among these MPAs is variable and dependent on latitude, species life history and habitat (Coleman & Kelaher, 2009; Coleman *et al.*, 2011b; Coleman, 2013). Here, we use oceanographic modelling to forecast how connectivity among this network of marine reserves is likely to be altered under forecast scenarios of climatic change. Specifically, for key benthic marine species (kelp and sea urchins; Bennett *et al.*, 2015) we compare connectivity between contemporary and future climatic scenarios with a focus on (i) the role of each MPA as a source or sink of propagule dispersal within the network, (ii) variability in dispersal among pairs of MPAs within the network and (iii) self-seeding within each MPA. For the first time, we also examine temporal variability in connectivity within a climate change context. Anticipation of such changes will greatly enhance the ability of this network of MPAs to maintain connectivity into the future by incorporating such knowledge into legislated reviews of MPA zoning arrangements.

Materials and methods

We investigated oceanographic connectivity among a network of marine parks (gazetted MPAs) spanning 8° of latitude along the coast of New South Wales, Australia (Fig. 1). This network comprised six marine parks each containing a subnetwork of numerous spatially separated 'no-take' marine reserves interspersed with partially protected and open fished areas (Kelaher *et al.*, 2014, 2015a). We considered connectivity between six marine parks and an additional region off Sydney at 34ºS which currently lacks a marine park. The six marine parks ranged in size from ~ 22 000 to ~98 000 Ha. Five marine parks are located along the mainland coast of SE Australia, at 29°S (Cape Byron Marine Park, CBMP; Kelaher *et al.*, 2015b), 30°S (Solitary Islands Marine Park, SIMP), 32°S (Port Stephens Great Lakes Marine Park, PSGLMP), 35°S (Jervis Bay Marine Park, JBMP) and 36°S (Batemans Marine Park, BMP). All of these sites are potentially within the influence of the East Australian Current which transports warm water (and species) from the tropics poleward. The sixth marine park is an island (Lord Howe Island) that lies at 31°S and ~600 km to the east of mainland Australia. This marine park was included in analyses because the EAC bifurcates most often between 30 and 32°S (Cetina-Heredia *et al.*, 2014), giving rise to the Tasman Front which flows eastwards towards Lord Howe Island. South of the EAC separation, the EAC extension and mesoscale eddies dominate the circulation along the coast influencing marine parks within this region.

We chose the two dominant habitat-forming species in NSW as key species to model changes in connectivity: kelp

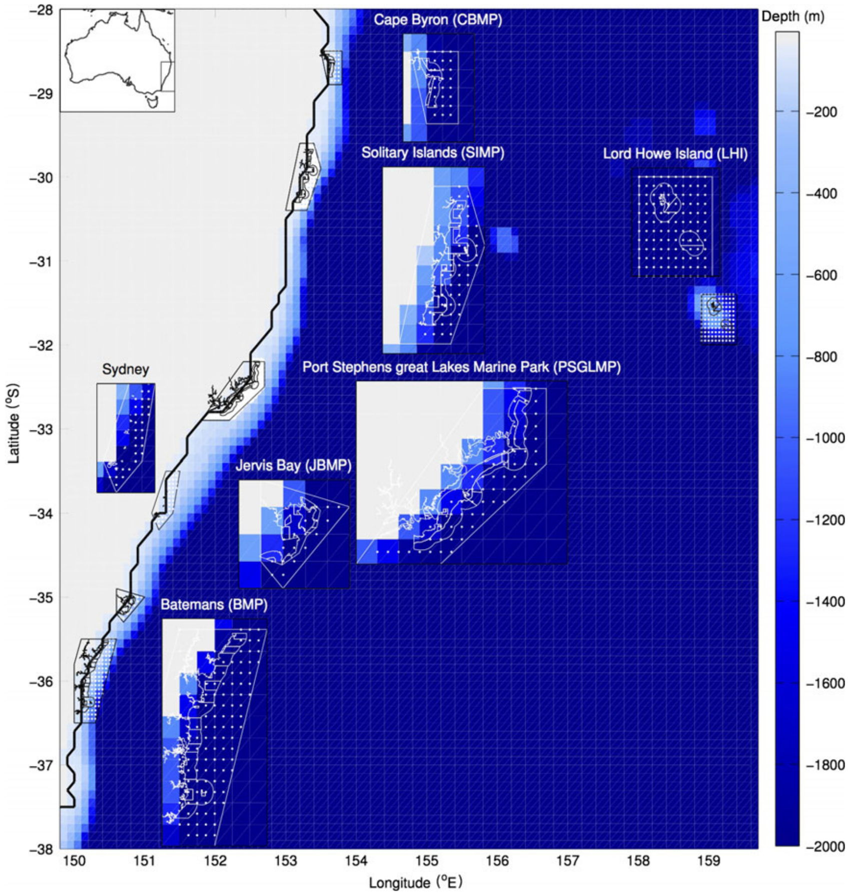

Fig. 1 Map of NSW showing a region off Sydney and networks of marine parks (along the coastline and zoomed in) used in this study. The horizontal distribution (latitude and longitude) of particle release locations (white dots) is also shown.

(*Ecklonia radiata*) and barrens forming sea urchins (*Centrostephanus rodgersii*; Andrew & O'Neill, 2000). These species were chosen because they play a disproportionately important (and interacting) role in structuring temperate marine biodiversity (e.g., Wernberg *et al*., 2003; Coleman *et al*., 2007). Moreover, these are key taxa in temperate marine communities that change due to protection (e.g., Shears & Babcock, 2003; Babcock *et al*., 2010; Coleman *et al*., 2013). Kelp does not occur on LHI so analyses of kelp connectivity were omitted at this site. *Ecklonia radiata* has a typical Laminarian alternation of generations life history strategy with large, perennial sporophytes (spore-producing individuals) alternating with microscopic gametophytes (gamete-producing individuals; Jennings, 1967). Significant dispersal likely occurs via zoospores and fertile sporophytes, which may disperse in currents when they are removed from the substratum during storms (Kirkman & Kendrick, 1997; Coleman *et al*., 2009). *Centrostephanus rodgersii* reproduces annually by gametogenesis and spawning (Byrne *et al*., 1998); thus, their geographical range is partly determined by larval dispersal. The onset of spawning occurs in mid to late June, and spawning is short (~1 month) at their northern limit but extends to 5 or 6 months at the southern edge of their range (Byrne *et al*., 1998). Based on metamorphosis of cultured larvae, Hugget *et al.* (2005) predict that settlement occurs from September to February. Nevertheless, settlement has been observed as late as early January (Andrew & Underwood, 1989). Hence, *Centrostephanus rodgesii* larvae take roughly 3–5 months to develop and become competent for settlement (Byrne & Andrew, 2013).

Current velocity outputs from the Ocean Forecast Australian Model (OFAM) forced with a contemporary and a future climate scenario were used to advect particles with the Connectivity Modelling Systems (CMS, Paris *et al*., 2013). The OFAM contemporary scenario is forced with climatology of present atmospheric conditions, while the future scenario is forced with the 2060 forcing, constructed from the climatology of present atmospheric conditions plus air-sea flux anomalies from the CSIRO Mk3.5 climate model under an A1B emissions scenario (Chamberlain *et al*., 2012; Sun *et al*., 2012). The A1B scenario considers rapid economic growth, global population that peaks in the mid-century and declines thereafter, and a balanced development of alternative energy technology (Nakicenovic *et al*., 2000). Model outputs capture the mean circulation and variability under contemporary and future atmospheric conditions but do not represent specific time periods. For this study, we use outputs from four representative 'years' of each scenario.

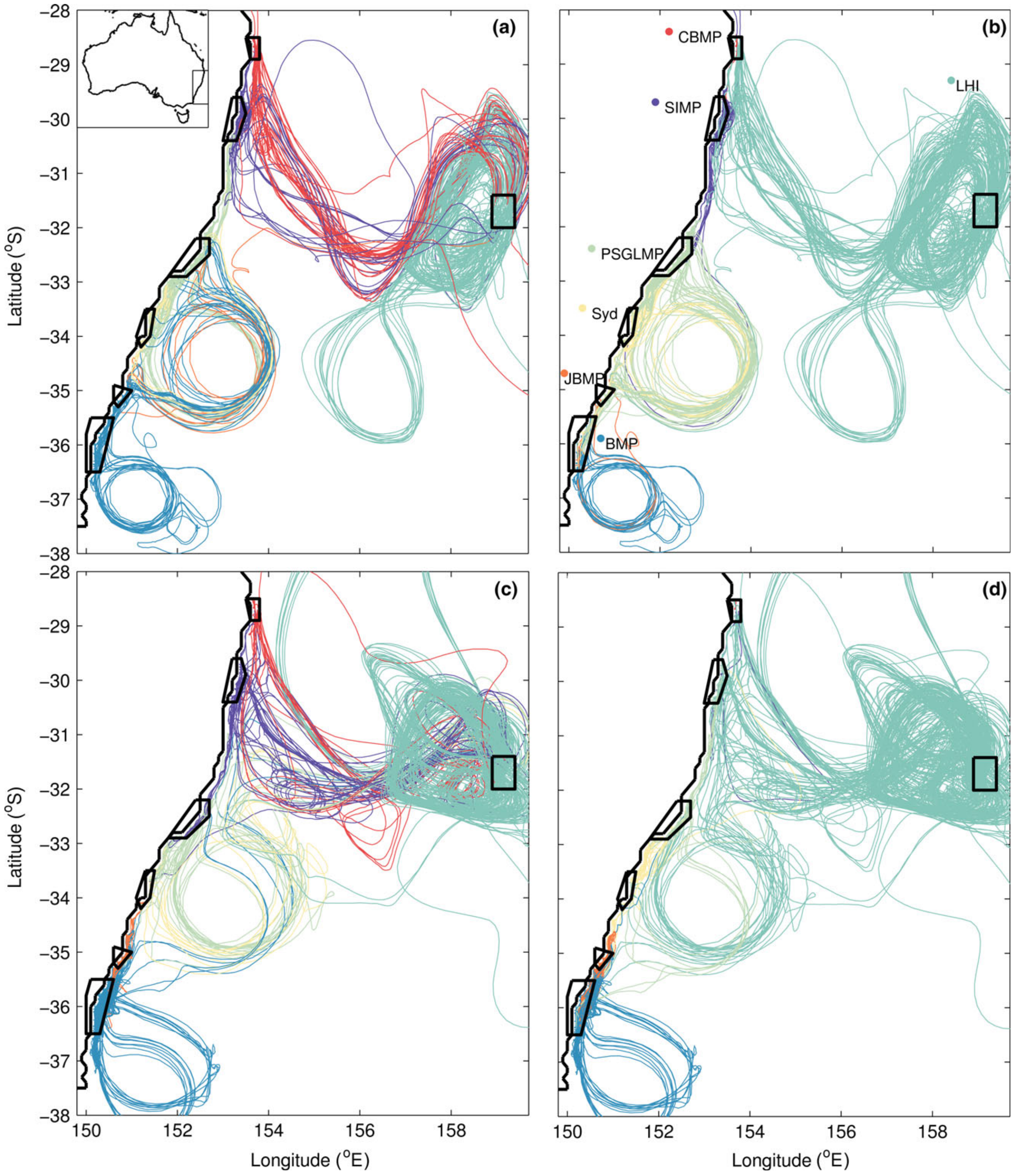

Fig. 2 Example of urchin particle trajectories that settled in Marine Parks for a contemporary year (a,b) and a future year (c,d). The trajectories are coloured by source (a,c) and sink (b,d) and show particles that achieve settlement.

Particle tracking simulations were conducted separately for each species, and particles were modified to reflect characteristics of *C. rodgersii* larvae and *E. radiata* sporophytes. We seeded particles inside the marine parks (Fig. 1) every 2 km in longitude and latitude, at 5-m depth and at 5-m increments from there to a depth of 25 m. Because marine parks are of differing sizes (areas), the number of particles released (each day) within each marine park varied from 47 to 600 and connectivity matrices correspond to settlement proportions relative to the total number of particles released inside each marine park. We released particles daily during the spawning season of *C. rodgersii* (June to October, Byrne *et al*., 1998) and the reproductive season of *E. radiata* sporophytes (April to October, Coleman *et al*., 2014), yielding a total of 1.20×10^6 and 8.55×10^5 released particles representing *E. radiata* and *C. rodgersii*, respectively, seeded over a 4-year period. Particles were considered to contribute to connectivity if they were within suitable habitat (i.e., inside an MPA) at any time 3–5 months after release (corresponding to the pelagic larval duration of *C. rodgersii*; Hugget *et al*., 2005) or between the release time and 2 months (corresponding to the maximal potential dispersal time of *E. radiata*; Coleman *et al*., 2014). Simulations were performed separately for four reproductive events (years) of each scenario and used to construct connectivity matrices (i.e., the number of particles exchanged among all pairs of marine parks, see Fig. 2 as an example of trajectories of particles exchanged among MPA's) for each year. The mean and standard deviation of the number of particles exchanged among marine parks were also computed for each scenario across years (Figs 4 and 5).

We compared differences between contemporary and future connectivity among all pairs of marine parks within the network. To do this, we subtracted contemporary from future connectivity matrices focusing on departures or changes from the contemporary state rather than in departures from a hypothetical future state. Subtractions were computed element by element of the connectivity matrix, where each element represents the exchange of larvae between a pair of marine parks within the network for each year. This yielded 16 matrices of change in connectivity (i.e., 16 possible pair combinations conformed using the four contemporary and four future years). To estimate changes in connectivity as percentages relative to total contemporary connectivity, the resulting matrices of change in connectivity were normalized (element by element) by the mean across years of the number of larvae settled in the contemporary scenario. Finally, the normalized matrices of change in connectivity were averaged to obtain a single matrix of changes in connectivity (Figs 4 and 5). Both poleward (south) and equatorward (north) contemporary and future connectivity were examined to determine whether there was asymmetry in the direction of any change.

To assess future changes in each marine park acting as a source (total dispersal into any other park in the network, minus self-seeding), acting as a sink (total dispersal from any other park in the network, minus self-seeding) and level of self-seeding within each marine park (the number of particles self-seeding), we did multivariate analyses in PERMANOVA using Euclidean distances. This was performed for each marine park separately with the factors being time (contemporary and

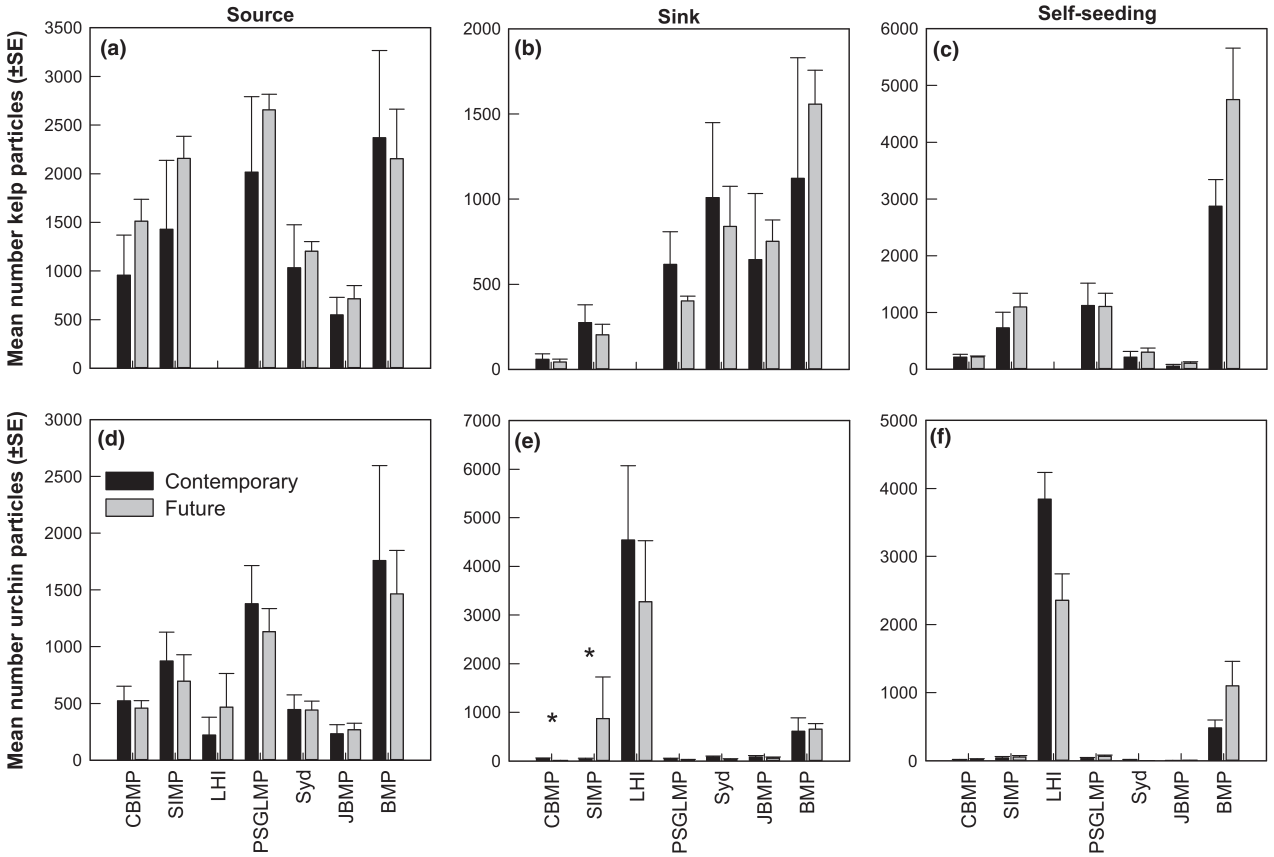

Fig. 3 Mean number of particles dispersing into the network from each marine park (source; a, d) and into each marine park from elsewhere in the network (sink; b, e) and self-seeding within each marine park (c, f) in the contemporary (black) and future (grey) climate scenarios for urchins and kelp.

future, fixed) and marine park source or sink (fixed, $n = 5$ and 6 for kelp and urchins, respectively) with $n = 4$ years of each scenario being the replicates. We were primarily concerned with the factor 'time' (see Fig. 3).

Results

Change in connectivity among populations of kelp

For kelp, connectivity among marine parks within the network varied greatly among years in both the contemporary and future scenarios, and this variation resulted in no statistical difference in each park acting as a source or sink between contemporary and future scenarios (PERMANOVA, Time: $P > 0.05$ for all parks, Fig. 3a, b). Either self-seeding of kelp particles within all marine parks did not change (CMBP and PSGLMP), or there was a trend for it to increase (all other parks, Fig. 3c and diagonal in Fig. 4). Self-seeding increased twofold in BMP and JB (Fig. 3c), but pairwise tests did not reveal any statistical difference despite a significant time × park interaction (PERMANOVA, $P > 0.05$ for all pairwise tests). Again, variation in self-seeding within each marine park was high in both future and contemporary scenarios (Fig. 3c and diagonal in Fig. 4).

The percentage and direction of change in connectivity among pairs of marine parks varied greatly, and variation among years was high in both the contemporary and future scenarios (Fig. 4). In general, there appeared to be a trend for poleward connectivity among pairs of marine parks to increase (reds, top right of matrix) and equatorward connectivity to decrease (blues, bottom left of matrix) (Fig. 4).

Change in connectivity among populations of sea urchins

For urchins, connectivity among marine parks within the network varied greatly among years in both the contemporary and future scenarios, and there was thus little overall statistical difference between scenarios in terms of marine parks acting as a source or a sink (PERMANOVA, Time: $P > 0.05$, Fig. 3d, e). The exception was the two lowest latitude marine parks (CBMP and SIMP) which received significantly less (CBMP) or more (SIMP) dispersal from elsewhere in the network under a future climate scenario (PERMANOVA, $P < 0.05$ for time × park interaction and for pairwise tests involving CBMP and SIMP). Self-seeding tended to increase marginally in most marine parks except LHI (Figs 3a and 5), but not

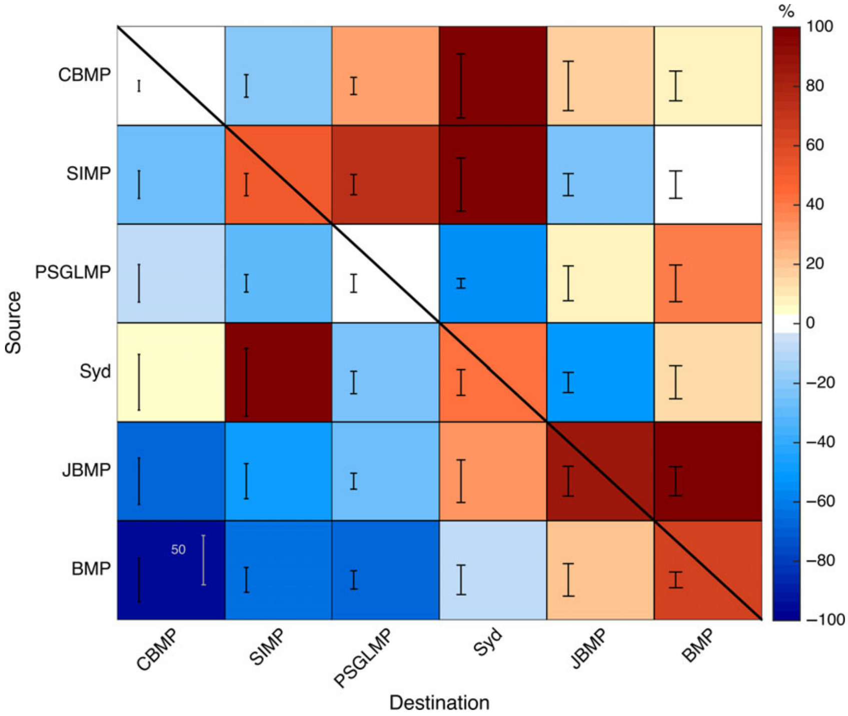

Fig. 4 Mean normalized difference (future minus contemporary) of connectivity among pairs of marine protected areas (MPAs) for *Ecklonia radiata*. Normalized connectivity difference is settlement proportion relative to the number of particles that settle in the contemporary scenario at each marine park. The error bars inside each square indicate the standard error across the normalized difference matrices for each pair of parks; a scale for the error bar magnitude is show in the bottom left of the figure. Both equatorward connectivity (bottom left of matrix) and poleward (top right of matrix) connectivity are shown.

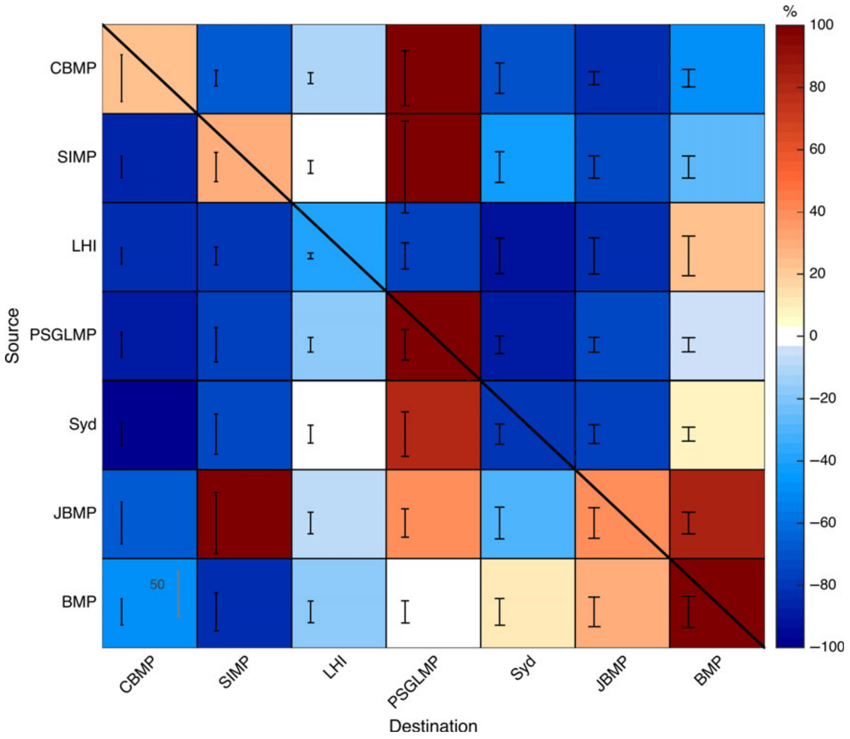

Fig. 5 Mean normalized difference (future minus contemporary) of connectivity among pairs of marine protected areas (MPAs) for *Centrostephanus rodgersii*. Mean normalized difference in connectivity is settlement proportion relative to the number of larvae that settle in the contemporary scenario at each marine park. The error bars inside each square indicate the standard error across normalized difference matrices for each pair of marine parks; a scale for the error bar magnitude is show in the bottom left of the figure. Both equatorward connectivity (bottom left of matrix) and poleward (top right of matrix) connectivity are shown.

significantly so (PERMANOVA, all pairwise tests for time × park interaction $P > 0.05$).

For sea urchins, connectivity among most pairs of marine parks tended to decrease in a future relative to contemporary climate scenario (Fig. 5), and this was the case for both poleward connectivity and equatorward connectivity (blues in Fig. 5). Interestingly, both future connectivity and contemporary connectivity into LHI were almost always extremely variable (Fig. 5) relative to all other parks.

Discussion

Planning the spatial distribution of MPAs to ensure they maintain connectivity both now and under future climatic scenarios is an important design criterion, but incorporating this objective is hindered by a lack of quantitative data (Magris *et al.*, 2014; Pendoley *et al.*, 2014), intricacies associated with species' ability to adapt (Mumby *et al.*, 2011), and biological responses (Andrello *et al.*, 2015) to climatic changes. Here, we use oceanographic modelling to quantitatively determine likely future changes to connectivity among a network of marine parks spanning over 1000 km of coastline. Climate-induced change in overall connectivity within the network generally did not exceed temporal variation within scenarios. However, despite temporal variation we found notable large and often asymmetrical changes to connectivity among specific pairs of marine parks and a tendency towards an increase in self-seeding. In the light of the nonsignificance of these patterns due to large variation in dispersal within each climatic scenario, they should be cautiously discussed in the review and design of spatial arrangements of marine parks.

Contrary to expectations, overall connectivity within the network of marine parks generally did not change significantly because variation among years within both contemporary and future climatic scenarios was great, making statistical analyses difficult. This great variation may have arisen due to our use of only 4 years of data for each climatic scenario, a limitation of computation time and data storage. Modelling capabilities that allow comparisons of more years under each scenario will be important for future studies to better understand temporal variability as well as enhance predictions about change. Predictions may be further refined and matched to spatial management initiatives by downscaling models under future scenarios to allow finer spatial resolution of likely changes to connectivity closer to the coast. In addition, incorporating information on variation in habitat availability, organismal behaviour and biological responses to climate change (e.g., larval survival and pelagic larval duration (PLD), Cetina-Heredia *et al.*, 2015) will also further refine predictions about future change to connectivity and provide the best possible scientific information to conservation managers.

Nevertheless, our study represents the first to examine temporal variability in connectivity among MPAs within both a contemporary and a future scenario and, therefore, to statistically examine future change. Given our finding of high temporal variability within climate scenarios, future studies should consider temporal variability to allow robust assessment when assessing absolute change, particularly given that temporal variation in dispersal is a key factor influencing population persistence (Blowes & Connolly, 2012). Even modest absolute change (3–10%) in overall connectivity metrics between contemporary and future climatic scenarios has previously been interpreted as meaningful in conservation settings (Andrello *et al.*, 2015), and may have genetic implications for populations. Absolute change in connectivity between many of the different pairs of marine parks studied here often far exceeded these estimates (up to 100% change), which should at least be considered (albeit cautiously) in the review and design of marine parks despite temporal variability within scenarios.

There were notable patterns in the intricacies of connectivity among pairs of marine parks, and these changes may have important implications for the geographical scale on which parks are reviewed. For kelp, poleward connectivity among most pairs of marine parks tended to increase. This is consistent with predicted strengthening of the EAC (Sun *et al.*, 2012), which likely facilitates longer distance transport of propagules among marine parks. Interestingly, the formation of more stable and longer-lived eddies is also predicted along south-east Australia (Oliver *et al.*, 2015), suggesting that increases to connectivity will not necessarily be linear in nature (stepping-stone; Kimura & Weiss, 1964) but that eddies may facilitate ‘jumps’ in dispersal. Increased poleward connectivity may have a number of genetic and demographic implications for kelp populations. If lower-latitude populations are better adapted to cope with warming temperatures, then increased connectivity may facilitate enhanced adaptation of populations at higher latitudes, which will experience increasingly warmer temperatures. For kelp (*E. radiata*), however, this may not be the case. Rather, low-diversity populations at low-latitude margins (Coleman *et al.*, 2011a) may lack the genetic diversity and physiological versatility to cope with future conditions (Wernberg *et al.*, 2016), and thus, increased poleward connectivity would not boost resilience of higher-latitude populations.

The trend towards an increase in poleward connectivity in kelp may be exacerbated by a general decrease in equatorward connectivity. This is particularly concerning given that lower-latitude kelp populations have already undergone climate-mediated range retractions (Smale & Wernberg, 2013; Verges *et al.*, 2014) which may be driven by lower genetic diversity (Coleman *et al.*, 2011a) conferring an inability to respond to thermal stress (Wernberg *et al.*, 2016), as well as increased herbivory (Verges *et al.*, 2014). The likelihood of lower-latitude populations recovering via recolonization from genetically diverse propagules at higher latitudes may thus be hindered under future climate scenarios resulting in continued poleward shifts in kelp distribution. Protecting important sources of equatorward connectivity in future marine park designs may thus be important for maintaining the resilience of kelp populations. Given that equatorward connectivity marginally increased (albeit nonsignificantly) only in adjacent networks of marine parks (e.g., BMP to JBMP, JMBP to SYD; Fig. 4), this could be achieved by ensuring the spacing among parks does not decrease and existing networks are maintained.

Similarly, for urchins, despite little change in overall connectivity within the entire network of marine parks, lower-latitude parks significantly changed their role as a sink in contrasting ways under a future scenario. Poleward and equatorward connectivity among pairs of marine parks showed a tendency to decrease in the future for urchins, and this is likely driven by the strengthening of the EAC (Oliver & Holbrook, 2014) and the long pelagic larval duration (PLD) of this species (3–5 months), resulting in more larvae transported to higher latitudes and outside the network of parks examined here. Although temperature-mediated decreases to larval PLD, dispersal distance and connectivity (e.g., see review by Lett *et al.*, 2010) may offset increased transport, this is unlikely to be the case for the urchin species studied here. Larvae of *C. rodgersii* have a wide thermal tolerance (Hardy *et al.*, 2014), and periodic incursions of warm, EAC water into higher latitudes have been shown to facilitate increased larval arrival, survival and establishment in cooler, higher-latitude areas (Ling *et al.*, 2008), contributing to the large reduction in abundance of giant kelp (*Macrocystis pyrifera*) forests (Johnson *et al.*, 2011). Thus, conservation managers in regions at higher latitudes will greatly benefit from understanding how the magnitude and spatial variability of urchin dispersal is likely to change under future climate scenarios, so that marine parks can be designed or reviewed to allow an increase in the resilience of important *M. pyrifera* forests that underpin biodiversity and local productivity. Although our study did not disentangle the effects of increased temperature on larval survival from concurrent increased poleward connectivity (but see Cetina-Heredia *et al*., 2015), it is likely that urchin larvae in northern NSW will begin to experience temperatures that exceed their thermal maxima (~22 °C; Hardy *et al*., 2014). Thus, although poleward transport in ocean currents may theoretically increase, this may be offset by a decrease in larval survival, particularly at lower latitudes and possibly a poleward range retraction once extant adult populations expire.

An important aspect of MPA design is the extent to which protected areas can be self-sustaining (Almany *et al*., 2007), particularly in the light of increasing habitat loss and fragmentation in surrounding unprotected areas. In the network of marine parks studied here, self-seeding (particle retention) of both species generally increased (albeit not significantly) in most parks and increased twofold for both species in higher-latitude marine parks. Such increase is likely a consequence of higher eddy activity, particularly polewards of latitudes where the EAC separates (Cetina-Heredia *et al*., 2015) where SYD, JBMP and BMP are located. The conservation implications of greater local retention are likely to be positive and confer population persistence (Figueiredo *et al*., 2014; Castorani *et al*., 2015). At lower latitudes where thermal stress is greatest, marginal increases in retention may mean that locally adapted genotypes are retained within protected areas. In addition, given that higher-latitude populations may have higher genetic diversity (kelp; Coleman *et al*., 2011b), greater self-seeding under future climatic scenarios likely confers long-term resilience and persistence of populations in these marine parks.

Ongoing management of MPAs following a review of the spatial arrangement and effectiveness of different management zones is a common requirement around the world to ensure MPAs achieve their goals into the future, but the extent to which this occurs is not clear. Anticipation of future changes to connectivity (and the variability surrounding predicted changes) will assist with such management processes by (i) identifying key additional areas to be considered for protection into the future and (ii) providing insight into the future effectiveness of current spatial arrangement of MPAs and how this may be best modified to maintain connectivity. Anticipating changes to connectivity, particularly for organisms that underpin biodiversity (e.g., habitat formers) and those that are key components of marine food webs, is important if MPAs are to achieve their conservation goals in a future of rapid environmental change.

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