Short communication

Recruitment hotspots boost the effectiveness of no-take marine reserves

Colin K.C. Wen [a,*], Glenn R. Almany [b], David H. Williamson [b], Morgan S. Pratchett [b], Thomas D. Mannering [a], Richard D. Evans [c,d], Jeffrey M. Leis [e], Maya Srinivasan [a], Geoffrey P. Jones [a,b]

[a] School of Marine and Tropical Biology, James Cook University, Townsville, QLD 4811, Australia
[b] ARC Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD 4811, Australia
[c] Department of Parks and Wildlife, 17 Dick Perry Ave., Kensington, Perth, WA 6151, Australia
[d] Oceans Institute, School of Plant Biology, University of Western Australia, Crawley, WA 6009, Australia
[e] Ichthyology, Australian Museum, Sydney, NSW 2010, Australia and Institute for Marine and Antarctic Studies, University of Tasmania, Hobart, TAS 7001, Australia

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ABSTRACT

No-take marine reserves are widely advocated as a means to conserve biodiversity and sustain fisheries. Within adequately protected reserves, the abundance, mean size and age of targeted fish and invertebrate species are often increased. However, the magnitude and rate of recovery within reserves varies among locations and species, and increases in abundance in reserves are not universal. One potential contributing factor to variability in reserve performance is spatial variability in the supply of recruits. Many reef fish species exhibit disproportionate levels of recruitment in relatively small areas (so-called "recruitment hotspots"). Here we tested the hypothesis that the presence of recruitment hotspots enhances the effectiveness of reserves for two important fishery species, coral trout (*Plectropomus maculatus*) and stripey snapper (*Lutjanus carponotatus*), on coral reefs of the Keppel Island group, Great Barrier Reef. To do this we compared fish densities in areas that did or did not contain previously identified recruitment hotspots, both inside and outside reserves. For both species, the mean densities of adult fish above the minimum legal size were approximately three times as high in reserves with hotspots than in reserves without hotspots. Furthermore, the mean densities in reserves without hotspots were similar to those in nearby areas that were open to fishing. In contrast, the densities of sub-legal size individuals of both species were primarily explained by the presence of recruitment hotspots, irrespective of reserve protection. Our results suggest that identifying and incorporating recruitment hotspots into the selection of sites for reserves could enhance both conservation and fisheries objectives.

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1. Introduction

Networks of no-take marine reserves are being implemented around the globe, with the primary objectives of protecting biodiversity and/or enhancing fishery sustainability (Gell and Roberts, 2003; McCook et al., 2010; Fenberg et al., 2012). Numerous studies have demonstrated that the abundance, mean body size and age of exploited fish species can increase dramatically within the boundaries of adequately protected reserves (Halpern and Warner, 2002; Lester et al., 2009; McLean et al., 2011). However, there is considerable variation among studies, both in the magnitude of abundance increases in reserves relative to fished areas (20–490% more fish inside reserves; Williamson et al., 2004; Abesamis et al., 2006; Harmelin-Vivien et al., 2008) and in the rates of increase after reserves are established (Russ et al., 2008; Russ and Alcala, 2010). A number of studies have also demonstrated little or no effect of reserves on the abundance of target fishery species

(i.e. Mapstone et al., 2008; McLean et al., 2011). The mechanisms by which some fish populations rapidly increase in abundance in reserves, and others do not, are poorly understood.

A range of factors may explain the variable success of reserves, including historic fishing pressure (Claudet et al., 2010), levels of compliance (Guidetti et al., 2008; Claudet and Guidetti, 2010; Pollnac et al., 2010) and the life history and mobility of target species (Halpern and Warner, 2003; Nardi et al., 2004; Claudet et al., 2010). Assuming that mortality from fishing is largely eliminated inside reserves, the reserve population can persist through time only if recruitment levels equal natural mortality, and for abundances to increase, recruitment must exceed natural mortality. In some cases abundance may increase in the reserve through immigration from nearby fishing areas, but this effect is likely to be significantly less important than recruitment over the long term (Stockhausen et al., 2000; Gerber et al., 2005). As a result, a key factor that contributes to the success of a reserve is whether or not it contains areas that consistently receive high levels of recruitment (Freeman et al., 2012). Despite the obvious importance of recruitment to reserve effectiveness, patterns of recruitment inside and outside reserves,

* Corresponding author. Tel.: +61 7 4781 5569; fax: +61 7 4725 1570.
E-mail address: colin.wen@my.jcu.edu.au (C.K.C. Wen).

and their relationship to the adult abundance of exploited fishes, have not been examined. Although the selection of sites for reserves must inevitably take into account a range of social and ecological factors (Fernandes et al., 2005; Beger et al., 2010; Edwards et al., 2010; Adams et al., 2011; Hansen et al., 2011), the inclusion of adequate recruitment and nursery habitats inside reserves may be an important ingredient for their long-term success (Kininmonth et al., 2011).

For most demersal marine fishes, recruitment occurs after individuals have survived the pelagic larvae stage, settled into benthic habitat and survived the first few weeks post-settlement to join the local population (Armsworth, 2002). Coral reef fishes typically exhibit predictable spatial patterns in recruitment, both within and among reefs (Valles et al., 2008). Reef locations that consistently receive above average levels of recruitment have been referred to as "recruitment hotspots" (Booth et al., 2000; Eagle et al., 2012; Wen et al., 2013a). Recruitment hotspots may be explained by a variety of processes and factors, including larval supply (Meekan et al., 1993) and the quality of juvenile habitat (Feary et al., 2007; Wen et al., 2013b). Many exploited fish species that are expected to increase in reserves also exhibit predictable spatial variation in patterns of recruitment (Leis and Carson-Ewart, 1999; Kingsford, 2009; Quéré and Leis, 2010). However, the role of recruitment in explaining the effectiveness of coral reef reserves has received little attention.

The aim of this study was to evaluate the hypothesis that well-connected reserve networks encompassing recruitment hotspots will be more effective than those established in areas that receive lower levels of recruitment. To test this, we sampled an existing network of reserves with a structured sampling design to compare the density of adults and sub-adults of two exploited fishes inside and outside reserves, in areas with and without recruitment hotspots. We focused on two commercially and recreationally important fishes, coral trout (Plectropomus maculatus) and stripey snapper (Lutjanus carponotatus). As these species are subject to minimum legal size limits in Australian waters, we also examined the effects of the 4 combinations of 2 factors (reserves and hotspots) on the size-frequency distribution of fishes.

2. Methods

2.1. Study location and species

This study was conducted during February 2010 on fringing coral reefs of the Keppel Island group (23°10′S, 150°57′E), an archipelago in the southern region of the Great Barrier Reef Marine Park. It focused on three no-take marine reserves that form part of a network of six no-take marine reserves in the Keppel Island group (Fig. 1). Two reef fish species were examined, bar-cheeked coral trout (Plectropomus maculatus; Serranidae) and stripey snapper (Lutjanus carponotatus; Lutjanidae), both important species in recreational and commercial fisheries on the Great Barrier Reef. Significant increases in the abundance and/or mean size of these species have been documented within no-take reserves of the Keppel Islands since 2004 (Russ et al., 2008; McCook et al., 2010).

2.2. Defining recruitment hotspots

We defined "recruitment hotspots" as areas that consistently receive higher than average recruitment (sensu Booth et al., 2000; Eagle et al., 2012). Suitable habitat for newly-settled juveniles (0–1 year-old) of P. maculatus and L. carponotatus have previously been found in nearshore, back reef habitats with abundant tabular and corymbose Acropora corals over sandy substrates (Kingsford, 2009; Wen et al., 2013a). Previous underwater visual census (UVC) data from reefs in the Keppel Islands has shown that juvenile densities in these specific habitats were much higher than in all other reef habitat types (Wen et al., 2013a).

To make an apriori selection of specific hotspot and non-hotspot sites for this study, we selected potential sites in both reserve and non-reserve areas. We compared catch per unit effort (CPUE) data based on diver collections of the two species carried out during two 20-day field trips in May 2008 and February 2009 (Table 1). Twelve divers collected juveniles of both species within an approximately 200 m-radius area from three boats, which were separated by at least 500 m to enable independence and to effectively cover the majority of the reef habitat area at each sampled location (Fig. 1). Data were standardized to catch per unit effort (number of juvenile fish collected per person per dive). Sites with distinctly high CPUE were defined as recruitment hotspots (Table 1). The thresholds of CPUE between hotspots and non-hotspots were >1.5 for P. maculatus and >2 for L. carponotatus. The reefs around Halfway Island and Clam Bay, which are split-zoning, were identified as recruit hotspots (see Table 1). These reefs have large areas of suitable recruitment and juvenile nursery habitat for these species. On the other hand, the reefs surrounding Middle Island, Humpy Island, Miall Island and Monkey Bay had similar suitable habitat, but less of it, and were defined as non-recruitment hotspots. The mean CPUE (±SE) across the four non-hotspot sites was 0.53 ± 0.11 and 0.91 ± 0.15 for P. maculatus and L. carponotatus respectively

2.3. Sampling design and survey methods

An orthogonal sampling design was employed to compare densities of fish in reserves with and without recruitment hotspots, and non-reserves with and without recruitment hotspots. This sampling design permitted analyses using two factors; (1) Zone (Reserve or Non-reserve), and (2) Recruitment hotspot (presence or absence). A total of 8 sites were surveyed using underwater visual census (UVC), 4 sites within no-take reserves and 4 sites in areas that were open to fishing. All the sampling sites had similar reef environments and 2 sites from both reserve and fished areas contained recruitment hotspots (Fig. 1). The reef area within each location was measured from satellite imagery using the software program ImageJ (v. 1.4.6; Rasband, 1997). The specific areas were (1) Eastern Clam bay (reserve with recruitment hotspot): 30.8 ha, (2) Western Clam bay (fished area with recruitment hotspot): 4.5 ha, (3) Western Halfway Island (fished area with recruitment hotspot): 13.2 ha, (4) Eastern Halfway Island (reserve with recruitment hotspot): 10.1 ha, (5) Humpy Island (fished area without recruitment hotspot): 33.5 ha, (6) Monkey bay (reserve without recruitment hotspot): 9.5 ha, (7) Middle Island (reserve without recruitment hotspot): 17.9 ha and (8) Miall Island (fished area without recruitment hotspot): 7.4 ha. The proximity of reserve and non-reserve hotspots may be considered a potential confounding issue for highly mobile species. However, biannual surveys over the past decade suggest strong consistent differentiation in mean densities between reserves and fished areas of the Keppel Islands (Russ et al., 2008; Evans and Russ, 2004). Furthermore, Plectropomus spp. and L. carponotatus are known to have relatively restricted home ranges (Zeller, 2002) and low rates of movement between reserves and fished areas are unlikely to significantly change the demographics of populations within those areas.

We recorded the number and estimated the total length (TL) of all sighted individuals of both species along 30-min timed-swim UVC transects. The length of each transect was measured using a diver-towed GPS system. The mean transect length was 112 m ± 17 m (SE). At each of the 8 sites, 6 replicate transects were conducted at randomly selected sites along reef flats and reef