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Population estimates and size structure of *Plectropomus leopardus* (Pisces : Serranidae) in relation to no-fishing zones: mark–release–resighting and underwater visual census

D. C. Zeller^A and G. R. Russ

School of Marine Biology and Aquaculture, James Cook University, Townsville, Qld 4811, Australia

^ACorresponding author. Present address: Fisheries Centre, University of British Columbia, 2204 Main Mall, Vancouver, BC, V6T 1Z4, Canada. email: Zeller@fisheries.com

Abstract. A mark–release–resighting (MRR) technique was used to estimate population size of the coral trout, *Plectropomus leopardus*, on coral reefs fringing Lizard Island, Great Barrier Reef, Australia. Fish were captured by hook-and-line fishing, and marked with individual freeze-brand numbers in August 1995. An underwater visual census (UVC) technique was used during September and October 1995 both for resighting of marked fish and to make an independent estimate of fish density and thus population size. The study area was 750 966 m². The UVC sampled 154 000 m² (20.5%) of this area. Six different methods of analysis of MRR gave similar population size estimates (e.g. Petersen 12 873; 95% CI 9989–15 754) extrapolated to the 4.5 million-m² reef area from datum to 20-m depth around Lizard Island. UVC gave a population size estimate (24 182; 95% CI 21 860–26 504) twice that of MRR. The lower estimate derived from MRR may be the result of tag-induced mortality, or of the relative difficulty in discriminating between marked and unmarked trout by UVC. This is only the second estimate of population size of coral trout on an area of the Great Barrier Reef.

Extra keywords: coral trout, fisheries, freeze brand, mark–release–recapture, marine protected areas, population size, visual census

Introduction

Estimates of abundance of fish populations are central to reliable assessment and management of fished species, and may assist in distinguishing between the effects of harvesting and natural changes in population size. Underwater visual census (UVC) surveys conducted along standardized transects have been used extensively to obtain abundance and density estimates of fishes in the marine environment, particularly on coral reefs (Brock 1982; Bellwood and Alcala 1988; Ayling and Ayling 1992; Russ and Alcala 1996). The technique has received detailed analysis in terms of precision, accuracy, bias and efficiency (Sale and Sharp 1983; Bell *et al.* 1985; Thresher and Gunn 1986; StJohn *et al.* 1990; Samoily 1997b). It has been reported that visual census surveys have provided precise and relatively accurate density estimates, especially for coral trout (*Plectropomus* spp., Cappo and Brown 1996), a major target of commercial and recreational line fisheries on the Great Barrier Reef, Australia (Kailola *et al.* 1993). Mapstone and Ayling (personal communication) assessed the optimal use of visual census of strip transects for estimating densities, and indicated that width of transects rather than length significantly affected estimates of density of coral trout. They concluded

that the effect of transect size was not caused by trained observers failing to count or over-counting fish moving in or out of the surveyed transects. Rather, the effect was thought to be due to systematic variation in the intensity with which observers searched transects (time taken per unit area). They recommended the use of 5-m wide transects for use with large, mobile fish.

In traditional fisheries research, mark–release–recapture studies are one of the methods most commonly used to assess population size, capture probabilities and mortality rates of target species (Ricker 1975; Nichols and Pollock 1983; Lebreton *et al.* 1992; Nichols 1992). In coral reef fisheries research, external tags of the anchor and dart types are the forms of tagging most commonly used (Recksiek *et al.* 1991). These tags are economical, small and easy to apply, and allow large samples of fish to be tagged. However, loss rates for external tags are high for coral reef fishes (Davies 1995), thus introducing a substantial bias in estimates derived from recaptures (Ricker 1975; Arnason and Mills 1981).

Hot or cold thermal marking techniques represent an alternative tagging method (Knight 1990; Hargreaves 1992; Light and Jones 1997; Samoily 1997a). Retention times of brands are highly variable and species specific, with a

maximum period recorded of 2 years (Hargreaves 1992). Thermal marking has been used with coral reef fishes only rarely (Light and Jones 1997; Samoily 1997a; Zeller and Russ 1998). Samoily found, on average, that brand retention was good for the first two months, with brands starting to fade thereafter. Unlike standard external tags, large individually numbered thermal brands permit visual identification of individually marked fish under water from a distance by divers using SCUBA (Samoily 1997a).

The tropical demersal reef line fishery on the Great Barrier Reef catches well over 125 species at some time (Mapstone *et al.* 1996). However, coral trout are the most heavily targeted species complex (five spp., with *P. leopardus* and *P. laevis* the major species) and form the bulk of the catch (35–45%, Mapstone *et al.* 1996). In 1994 the reported commercial catch for coral trout was ~1280 t (Mapstone *et al.* 1996), with a value of \$A8.2 million in 1989–90 (Kailola *et al.* 1993). Coral trout are also of major importance in the large recreational fishing sector, with the quantity of coral trout estimated to be at least the same as, or more than, the commercial catch (Kailola *et al.* 1993).

The suitability of underwater visual census surveys for assessing coral trout populations, and the satisfactory levels of underwater readability of cold brands reported for *P. leopardus* (Samoily 1997a), provided a unique opportunity to use these two methods in a novel approach to mark–release–resighting. This approach had the following aims: (i) to provide an estimate of population size and (ii) to evaluate the size structure of *P. leopardus* at Lizard Island. This involved use of a standardized mark–release–resighting technique. The study combined initial capture by hook and line with a fisheries-independent resighting method (underwater visual census). The population estimates obtained through the mark–release–resighting study were evaluated in relation to the existing marine protected area (no-fishing) zoning at the study site.

Methods

Study site and sampling protocol

This study formed part of an investigation into patterns of movement of coral trout (Zeller 1997, 1998; Zeller and Russ 1998), and was undertaken on the sheltered western and northern sides of Lizard Island (Fig. 1), northern Great Barrier Reef, Australia (14°40'S, 145°28'E). The reef area around Lizard Island is divided into management zones with areas open or closed to fishing (duration of closure, 12 years; Fig. 1).

Details of the capture, marking and visual census methods are described in Zeller and Russ (1998). In summary, capture of fish for external marking was by standard commercial reef line-fishing techniques undertaken in August 1995 by two commercial fishers working independently from small aluminium dinghies. The spatial and temporal patterns of fishing effort were distributed evenly around the north, north-east and western sides of the island (Fig. 1). The maximum fishing depth was restricted to 10 m to facilitate recapture by visual census. Captured specimens were processed by the senior author, as follows. Fork length (FL) of each fish was measured, the fish were double tagged with standard T-bar anchor tags (Hallprint, Australia), and individual freeze-brand numerals were applied on each side of the caudal peduncle, prior to release of the fish at the capture site (Fig. 2).

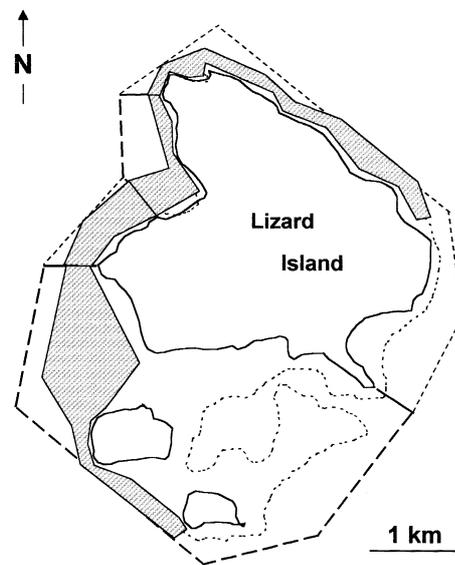


Fig. 1. Study area with the Great Barrier Reef Marine Park management zones. Hatched area: reef area used for the study (patch and fringing reef habitats, datum to 10-m depth). Dotted line: approximate reef outline. Short-dashed lines: zones open to fishing. Long-dashed lines: zones closed to fishing (reserves). Solid lines: reserve boundaries.



Fig. 2. Photo of *P. leopardus* (FL 43.3 cm) carrying freeze-brand number 151, viewed under water. Standard T-bar anchor tags, near anterior end of dorsal fin on either side of the fish, are also visible. (Photo courtesy of R. Grace and TV New Zealand.)

'Recapture' of marked animals was by resighting during underwater visual strip-census surveys using 100 × 5-m transects. The transect length of 100 m was chosen to minimize the occurrence of zero counts of target species (A. M. Ayling, personal communication). Choice of width of transect (5 m) was based on the minimum underwater visibility encountered around Lizard Island and on the recommendation by Mapstone and Ayling (personal communication). Transect tapes were laid while visual surveys were being conducted, to reduce attraction of, or disturbance to, the fish by previously laid tapes (Fowler 1987). Censuses were conducted between datum (lowest astronomical tide) and the 10-m depth contour (or sand–reef interface), with the average census depth being 6–8 m. Two separate resighting surveys were con-

ducted during September (Zeller) and October (Zeller and Russ) 1995, being one and two months after the initial capture, respectively. Total sampling effort was 167 transects (83 500 m²) in September and 141 transects (70 500 m²) in October, covering a total area of 154 000 m². A previous pilot study indicated that numerical freeze brands remained readable under water for three months (Zeller 1996). During each census, all coral trout sighted within the transect width and up to 5 m in front of the projected transect line were counted, and their sizes (total length) were estimated in 5-cm size classes. Before each survey, the method of Bell *et al.* (1985) was used to train observers in size estimation. Sightings of tagged fish were recorded and visible freeze brands were identified. When a fish with freeze brands was re-sighted and its length was visually estimated, comparison with the length measured during the freeze-branding process constituted an ongoing verification of the accuracy of visual estimation; any potential growth increments since tagging would not have exceeded the 5-cm size groups used for allocation.

Data analysis

The estimates of population and size structure were evaluated with respect to management zones (open v. closed to fishing), and time interval between recaptures. The time intervals were as follows: 1 month between line capture and first visual census (August–September UVC1); 2 months between line capture and second visual census (August–October UVC2); and 1 month between first and second census (UVC1–2). Statistical analyses comprised *t*-tests, Mann–Whitney *U*-tests, and χ^2 homogeneity and goodness-of-fit tests.

Two separate techniques were used to estimate population size—firstly, mark–release–resighting, and secondly, density estimates obtained from visual surveys. The mark–release–resighting estimates calculated were: the Joint Hypergeometric Maximum Likelihood Estimator (JHE, Neal *et al.* 1993), the Minta–Mangel Bootstrap Estimator (MM, Minta and Mangel 1989), the Bowden Model Estimator (BM, Bowden 1993) and the traditional mark–recapture estimators of Petersen (Ricker 1975), Bailey (1952) and Chapman (1952). Multiple resightings of individuals per survey were rare and not incorporated in the analyses. The first three estimates (JHE, MM and BM) were derived specifically for ‘resighting’ rather than for ‘recapture’ (White 1996). Hence, the initial sample of animals is captured and marked, but ‘recaptures’ are obtained by observation, not by physical recapture. Population size estimates were calculated from mark–release–resighting estimates by using NOREMARK (White 1996), mark–release–recapture formulae in Ricker (1975), and estimates of density derived from visual census coupled with area estimates of available reef substratum (patch and fringing reef habitats). The required reef area measures were digitized from calibrated aerial photographs with overlaid depth contours.

Results

In total, 183 coral trout were released after being marked with freeze brands. Of a total of 817 *P. leopardus* counted on transects (UVC1, 440; UVC2, 377 fish), 44 (UVC1) and 26 (UVC2) were resightings of branded specimens (Zeller and Russ 1998). The total hard substratum reef area as defined for the censuses (datum to 10-m depth) in the whole study area was 750 966 m² (Fig. 1), with the two surveys in September and October sampling a total of 154 000 m² (20.5%).

Population size estimates

All population estimators using mark–resighting data returned very similar results, with population estimates ranging from 1890 (BM, 95% CI 1527–2338) to 2134 (MM,

95% CI 1969–2242) coral trout for the total reef area (datum to 10-m depth) in the study area (750 966 m², Table 1, Fig. 3). From the average density estimate of obtained from the surveys (5.31 ± 0.26 fish 1000m⁻² (mean \pm s.e.m.), Zeller and Russ 1998), the population size estimate derived from UVC data for the same area was 3988 fish \pm 383 (95% CL, Table 1, Fig. 3).

On the basis of an assumption that the distribution and density of coral trout recorded in the patch and fringing reef areas of the study area were representative of the datum-to-10-m stratum for all patch and fringing reef areas of Lizard Island reef (shallow reef flats and lagoonal areas were excluded from all assessments), the population estimates were scaled up to the available reef area of 2 503 809 m² (Table 1). This resulted in population size estimates ranging from 6301 (BM, 95% CI 5091–7795) to 7115 fish (MM, 95% CI 6565–7475) for mark–resighting estimates, and 13 295 (95% CI 12 018–14 572) fish based on visual census (Table 1, Fig. 3). Further, assuming that density and distribution of coral trout for the stratum between 10 and 20 m were similar to those between datum and 10 m, the population estimates for mark–resighting and UVC ranged from 11 461 (BM, 95% CI 9260–14 177) to 24 182 (UVC, 95% CI 21 860–26 504) fish over a reef area of 4 553 331 m² (datum to 20-m depth, Table 1, Fig. 3).

Size distributions

The size frequency distributions of fish encountered during the study did not differ significantly between management zones open and closed to fishing ($\chi^2_{[11]} = 11.91$, $0.5 > P > 0.25$, Fig. 4a, 4b). The average size of fish recorded in open zones (FL 37.9 cm) did not differ significantly from that of fish recorded in closed zones (FL 39.1 cm) ($t_{[1040]} = 1.6015$, $P = 0.1096$). However, the size distribution for fish collected by hook and line differed significantly from that obtained by visual census ($\chi^2_{[11]} = 43.62$, $P < 0.001$, Fig. 5a, 5b). The average size of fish collected by hook and line (42.9 cm) was larger (Mann–Whitney $U_{[216,826]} = 6.1129$, $P < 0.001$) than that recorded by visual census (37.8 cm).

Discussion

Estimation of population size

The two dominant reef habitats (patch and fringing reefs) were not treated separately because average trout densities did not differ significantly between these two habitats (Zeller and Russ 1998), and initial capture effort was distributed evenly. Extensive areas of shallow reef flats and the large lagoon at Lizard Island were excluded from the present study. Since dive restrictions made it impractical to efficiently sample depths >10 m, neither initial fishing nor subsequent resightings were undertaken for this depth stratum. Extrapolation of the estimates from the censused depth stratum (datum–10-m) to the 20-m depth contour was based on the assumption that average densities of coral trout were

Table 1. Estimates of population size (with 95% CI) of *P. leopardus* at Lizard Island, northern Great Barrier Reef, Australia

Estimators: JHE, Joint Hypergeometric Maximum Likelihood Estimator; MM, Minta–Mangel Bootstrap Estimator; BM, Bowden Model Estimator; PET, Traditional Petersen Method; BAIL, Bailey Estimator; CHAP, Chapman Estimator; UVC, direct visual estimate

Reef area (m ²)	Reef section, depth	JHE	MM	BM	PET	BAIL	CHAP	UVC
750 966	Study area, datum–10 m	2116 (1750–2618)	2134 (1969–2242)	1890 (1527–2338)	2123 (1647–2598)	2096 (1633–2558)	2106 (1918–2294)	3988 (3605–4371)
2 503 809	Lizard Island, datum–10 m	7055 (5835–8729)	7115 (6565–7475)	6301 (5091–7795)	7078 (5492–8662)	6988 (5445–8528)	7022 (6395–7648)	13 295 (12 018–14 572)
4 553 331	Lizard Island, datum–20 m	12 831 (10 612–15 875)	12 940 (11 940–13 595)	11 461 (9260–14 177)	12 873 (9989–15 754)	12 709 (9902–15 510)	12 770 (11 629–13 909)	24 182 (21 860–26 504)

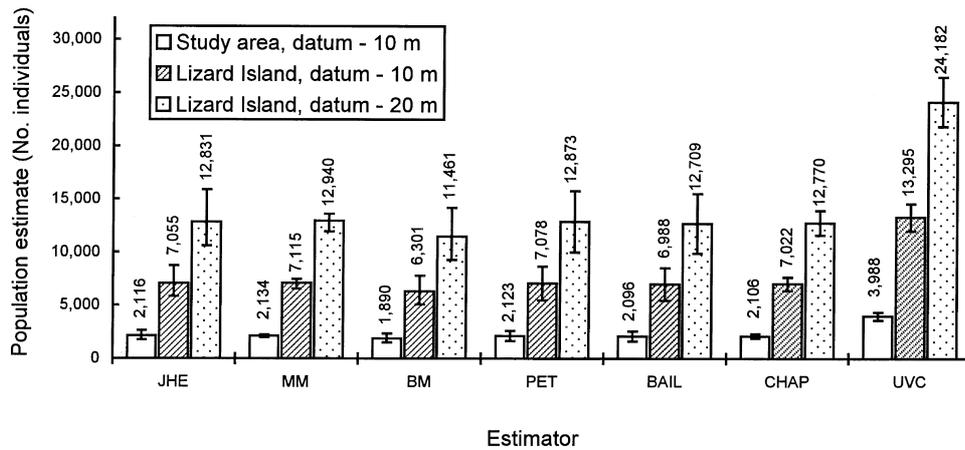


Fig. 3. Population size estimates for *P. leopardus* as derived from mark–release–resighting estimators and direct density estimation. Estimation models used: JHE, Joint Hypergeometric Maximum Likelihood Estimator; MM, Minta-Mangel Bootstrap Estimator; BM, Bowden Model Estimator; PET, Traditional Petersen Method; BAIL, Bailey Estimator; CHAP, Chapman Estimator; UVC, population estimate based on coral trout density obtained from visual surveys. Error bars are 95% confidence intervals.

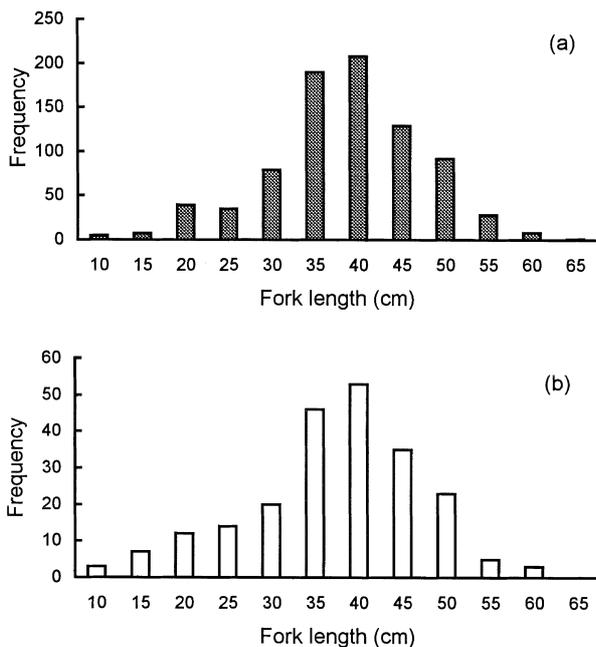


Fig. 4. Size distribution of *P. leopardus* at Lizard Island: (a) fish from zones closed to fishing; (b) fish from zones open to fishing.

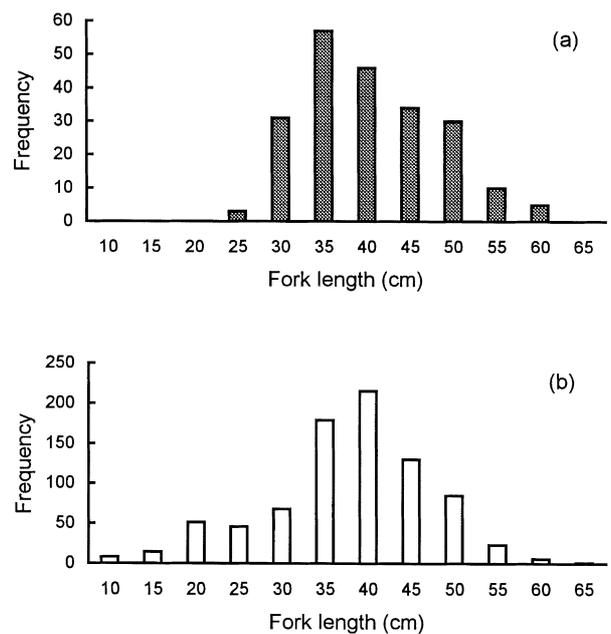


Fig. 5. Size distribution of *P. leopardus* recorded at Lizard Island (a) from hook-and-line catches and (b) by visual census.

the same for the deeper areas. This assumption seems reasonable, since Mapstone and Ayling (personal communication) found that coral trout densities were not depth-stratified, at least to depths of 20–30 m.

For coral reef fishes there are few examples of estimates of abundance or population size that have used mark–release–recapture (Appeldoorn 1996). Recksiek *et al.* (1991) used fish traps and an estimation technique for sparse capture–recapture data (Chao 1989) to estimate local abundances of four species of reef fish on a shallow patch reef in

Puerto Rico. Only one other study has attempted to estimate population size of coral trout on an individual reef of the Great Barrier Reef. Beinssen (1989) used the Petersen method on 83 tag recaptures over a 14-day period from 375 fish initially tagged in a sample collected by line fishing; he estimated the population size of coral trout on the Boulton reef (3 420 000 m², southern Great Barrier Reef) to be 8613 ± 873 (mean ± s.e.m.). This estimate compares well with the Petersen estimate of 12 873 fish obtained for the 4 553 331 m² reef area in the present study.

The comparison of population size estimates obtained from UVC density and mark–release–resighting estimators indicated that the UVC density estimate was between 1.87 to 2.11 times higher than those based on the resighting of freeze-branded specimens. In a review of the use of both UVC and mark–release–recapture techniques for estimating abundance and density of coral reef fish, Cappo and Brown (1996) concluded that UVC is a precise way of assessing coral trout density. Clearly, all UVC estimates have the potential to count individual fish more than once. Equally, it is also possible that animals that are present during a census are missed (the over- and under-counting effect examined by Mapstone and Ayling, personal communication). On the other hand, mark–release–recapture techniques, although used effectively for estimation of survival rates (Burnham *et al.* 1987; Lebreton *et al.* 1992), age and growth validation (Ferreira and Russ 1992, 1994; Davies 1995) and movement studies (Hilborn 1990; Schwarz and Arnason 1990; Hilborn and Walters 1992; Schweigert and Schwarz 1993; Davies 1995; Samoily 1997a; Zeller and Russ 1998), were considered to be less reliable for estimation of population sizes for coral reef fishes (Cappo and Brown 1996). Kearney (1989) lists six major assumptions underlying the use of tagging for population size estimation—firstly, the assumption of unchanged behaviour of tagged animals. Although this could not specifically be addressed in the present study, no unusual behaviour was observed during visual censuses, nor was any change in behaviour recorded during aquarium trials of the marking method used (Zeller, unpublished). This is further supported by the observation that freeze-branded fish participated at spawning aggregation events (Zeller, personal observation). Secondly, complete mixing of tagged fish and a uniform distribution over the total population is assumed. This assumption could not be tested, but given that the initial sampling distribution was determined by fishers' choice within an even distribution of overall fishing effort, uniform distribution can be assumed. Thirdly, it is assumed that recovery effort is uniform and covers the total population. Our survey sampling effort was uniformly distributed over the complete study area, and, as far as it was possible to ascertain, covered the total population within the study area. We acknowledge that the extrapolation to areas outside the study area (complete Lizard Island reef excluding lagoon and shallow reef flat, datum to 20 m depth strata) does not necessarily meet the assumptions. Fourthly, constant and known rates of immigration and emigration are assumed. These rates are not known. However, previous studies have indicated that coral trout are relatively sedentary and inhabit limited home ranges that are stable over periods exceeding the duration of the present study (Zeller 1997). This behaviour changes during the annual spawning season (Lizard Island peak season: October–November), when individuals move to aggregation sites for spawning before returning to their home ranges (Zeller 1998). The lower total

fish count (440 v. 377) and resightings of marked coral trout (44 v. 26) during the second census (October) could have been influenced by spawning behaviour. All known aggregation sites for coral trout at Lizard Island are in depths of >15 m (Zeller 1998), and were not included in the census. The fifth assumption relates to reliable and unbiased returns. Given that re-sightings were made by the authors, this assumption is believed to hold. The final major assumption, that size of total catch is known, holds for the same reasons.

Hilborn and Walters (1992) recommended that tagging studies should attempt to tag at least 25% of the population to estimate abundances. With our population estimates from the study area, this equates to between 470 (BM estimate of 1890) and 997 (UVC estimate of 3988) coral trout that would have had to be tagged. We acknowledge that the existing tagging effort of 183 coral trout is low, but, given the paucity of such population estimates for coral reef fisheries species, our estimate may serve as a useful guide for future assessments. Given that density estimates derived through visual census are sometimes found to underestimate true fish abundances, at least for abundant, shy or cryptic species (Stone *et al.* 1979; Brock 1982; St John *et al.* 1990), our visual census-derived estimate (24 182 coral trout) could be considered a more reliable, albeit potentially conservative, estimate of population size of *P. leopardus* on the reef around Lizard Island.

Size frequency distribution

No difference in size frequency distributions of *P. leopardus* were detected between management zones open and closed to fishing. This suggests three possibilities: that extensive movements of fish occur along the reef and therefore between management zones; that fishing effort is very low in zones open to fishing around Lizard Island; or that significant illegal fishing occurs in zones closed to fishing. Extensive tagging (Davies 1995; Samoily 1997a; Zeller and Russ 1998) and ultrasonic telemetry studies (Zeller 1996, 1997, 1998) indicate that movements of adult coral trout are limited, except during the reproductive season (Samoily and Squire 1994; Zeller 1996, 1998). The mean daily distance moved by coral trout within their home ranges (as measured by ultrasonic telemetry) was 192.2 ± 5.09 m (mean \pm s.e.m.), with a maximum of 1121.8 m (Zeller 1997). Furthermore, fishing pressure around Lizard Island is considered low and consists primarily of limited recreational fishing (L. Vail and A. Hoggett, personal communication). Infringements of existing fishing zoning rules are unlikely to occur on a regular basis at Lizard Island, because of the proximity and high activity of the local research station and resort. Thus, the similar size–frequency distributions between management zones could be accounted for by the low fishing effort at the study site. However, the evidence for the effects of fishing on the size structure of populations of coral reef fishes is relatively strong (review by Russ 1991).

Most studies of coral trout detected larger average sizes on whole reefs closed to fishing (review by Williams and Russ 1994). However, Ferreira and Russ (1995) found no significant difference in mean size of *P. leopardus* between reefs open and closed (for up to 7 years) to fishing in the central Great Barrier Reef. They attributed this contrasting result to high variability among replicate reefs, and indicated the importance of increased replication of reefs for improved statistical power. Large differences in apparent fishing pressures among replicate reefs open to fishing may have contributed also to their results, and variation in effort would need to be taken into account in future studies.

Mean size of coral trout captured by hook and line in this study was larger than that obtained from UVC. Coral trout <20 cm were not represented in hook-and-line catches, despite being sighted during UVC. Large specimens, on the other hand, appeared to be sampled with equal efficiency by both methods. Thus, sampling of coral trout using commercial hook-and-line methods resulted in a skewed size–frequency distribution, with under-representation of small size classes. The observed size selectivity between the two methods may also partially explain the difference in population estimate between UVC and mark–release–resighting estimates; although hook-and-line sampling appears to provide a good representation of population abundance for fish of harvestable size, it clearly underestimates smaller size classes for whole-population estimation. A similar size selectivity of hook-and-line capture was also reported by Strachan-Fulton (1996), who constructed a gear-selectivity curve for *P. leopardus* that demonstrated a minimum size of 23 cm available to hook-and-line capture. Similarly, D. J. Welch (personal communication) has found that a lower proportion of *P. leopardus* in the smaller size classes (<20 cm) could be collected by spear fishing (using a random selection protocol) than could be observed by UVC. Clearly, underwater visual census provides a useful tool for evaluation of size selectivity of hook and line and other extractive sampling techniques for coral reef fisheries, at least for non-cryptic or non-shy species (Kulbicki 1988).

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