

DISCUSSION

We have reported here a comprehensive survey of reefs in the Cairns Section of the GBRMP. Despite surveying 50 reefs, however, we found no clear evidence of an emerging outbreak of crown-of-thorns starfish (*Acanthaster planci*), with the numbers of individuals seen similar overall to those seen in the same region in the previous year (1990) (Mapstone et al. 1998a). It was perhaps noteworthy that most of the *A. planci* we saw were small-medium sized, but apart from at Endeavour Reef, they were uncommon. The emergence of boom populations of *A. planci* since 1994 (Engelhardt 1997; Engelhardt and Lassig 1996) suggests either that those individuals present in the boom populations were not present in 1991, or that they were small and sufficiently cryptic that non-destructive visual surveys would not have detected them.

The inter-annual variation in abundances from 1990 to 1991 we report suggest that monitoring the effects of management strategies will require considerable investment in longitudinal data to be sensitive to effects of management in the context of natural and/or observational variation. For most taxa, abundances differed substantially between years, though the origin of such variation cannot be certain. Given the expected longevity of many of the fishes (Brown 1994; Brown et al. 1993, 1996; Fowler 1990; Ferreira and Russ 1995; Mapstone 1988; Russ et al. 1996; Williams 1997; Choat unpub. data) and the multi-species nature of most measures of coral abundances, one- or two-fold real changes in abundance from year-to-year seem unlikely to be typical. Accordingly, it seems likely either that one or both of the years we considered was aberrant, or that the apparent differences reflected sampling variation and/or confounding of observers with survey.

The passage of cyclone Ivor through the Cairns Section of the Marine Park in mid-March 1990 (Done et al. 1993; van Woesik et al. 1991), after most of the data for that year had been collected, may represent an unusual event. Although cyclones *per se* are not unusual, the amount of habitat damage caused by cyclone Ivor was extreme on several reefs and apparently more than usual for cyclonic disturbances (Done et al. 1993, van Woesik et al. 1991). Such damage might have precipitated the large differences we saw between 1990 (pre-cyclone Ivor) and 1991, 12 months later. The relatively high frequency of counts of sessile benthos that were less in 1991 than in 1990 would be consistent with such an hypothesis. The greater counts of many of the fishes in 1991 compared with 1990 would be less conspicuously consistent with an hypothesis of cyclone disturbance, however, unless the habitat damage from the cyclone had increased visibility of some of the more secretive species.

The possibility of changed observer bias precipitating high inter-annual variation cannot be ruled out since, for both the fish and benthic taxa, observers in 1991 were not the same as those in 1990. Whilst this is clearly not desirable, it is likely to be a facet of most long-term monitoring programs. Since we were unable to train observers from one year against those from the previous year, however, the variation we saw might be expected to represent the worst case of change in observers that will have to be managed in long-term monitoring. Thompson and Mapstone (1997) considered the implications of observer-related bias and variation for long-term monitoring and, despite careful training of observers, they found substantial residual variation between observers for several taxa of reef fish. Such operational problems with long-term monitoring procedures indicate that if longitudinal monitoring for management strategy evaluation is commissioned, it should probably be commissioned through an institutional facility where multiple observers are available, adequate training of them is properly managed, and where the effects of staff-turnover can be minimised.

The results we present here do not indicate a clear, unequivocal set of patterns in abundances of several organisms related to either zoning history or notional tourist use. For many taxa, our sampling was insufficient to detect whatever effects of either past zoning or tourist use might have occurred. For other taxa, the zoning-related patterns frequently depended on either where across the continental shelf we looked, on the habitat considered, or on the notional history of consistent tourist use. Similarly, patterns in abundances related to frequency of tourist use also varied with habitat, zoning status, or shelf position.

Abundances of organisms were greater on non-tourist reefs than on tourist reefs in 12 cases, whilst the reverse was true in nine cases. If either result ($TS > NT$ or $NT > TS$) was considered equally likely, this arrangement (or one more extreme) of 21 cases in which a significant difference occurred would have a probability of 0.33. At face value, therefore, it seems unlikely that the effects of tourist use were manifest predicably as either increasing or decreasing abundances of biota.

Despite the heterogeneity in effects of zoning history, it was more often the case that MNP reefs had significantly greater abundances of organisms than GU reefs (23 cases) than the reverse ($GU > MNP$ – 11 cases). If it was considered equi-probable that MNP reefs or GU reefs would have the greater abundances of organisms, then in a set of 34 differences between GU and MNP reefs approximately 17 would be expected to show greater abundances on MNP reefs and 17 would be expected to show greater abundances on GU reefs. The probability that GU reefs would be greater than MNP reefs in only 11 (or fewer) of the 34 contrasts by chance alone would be only 0.029. In most cases, greater abundances on MNP reefs than on GU reefs would be interpreted as evidence of a desirable effect of protection from exploitative use (protective zoning). Such an inference rests on the premise that greater abundances are desirable, and that the lack of protection from some uses on GU reefs had resulted in declines in abundances of some organisms.

Whilst it may be tempting to infer that this is evidence of an effective zoning strategy, such an inference is weak and should be regarded with great caution because of the limited information from which it arises. Mechanisms by which such declines might have been caused are not obvious for some of the organisms for which effects were apparent, and for others that are known to be targets of harvest (e.g. some emperors, snappers, and coral trout—Mapstone et al. 1996a), no effects of zoning history were apparent. In some cases the lack of effects may be at least partly related to the inappropriateness of underwater visual survey for counting those species that are most targeted by fishing (e.g. *L. miniatus*, *L. nebulosus*) (Mapstone and Ayling 1998), or because many species are not generally targets of a fishery or collection activity (e.g. several of the lutjanids and lethrinids) (Mapstone et al. 1996a). Coral trout (*Plectropomus* spp.), however, are fairly amenable to underwater visual survey (Mapstone and Ayling 1998) and are the fishes most targeted by the reef line fishery (Higgs 1996; Trainor 1991; Mapstone et al. 1996a).

The absence of effects of zoning history for many of these species could reflect several alternative situations. First, it may be that fishing pressure on GU reefs in the Cairns Section has been relatively ineffective in reducing numbers of target species such as coral trout. The gross levels of commercial reef line fishing effort and catch over most of the Cairns Section apparently has been low relative to elsewhere on the GBR (Mapstone et al. 1996a). On the other hand, Blamey and Hundloe (1993) reported that recreational fishing on the GBR in the Cairns region was greater than elsewhere. There are insufficient data available, however, to describe the patterns in recreational fishing over the whole of the Cairns Section, although it seems likely that recreational reef line fishing would be more common in the southern half of the Section simply because of the greater coastal settlement south of Mossman (Higgs 1996). Alternatively, reductions in numbers of larger coral trout (through fishing) might have been compensated by increased numbers of smaller coral trout, most likely because of increased survivorship of juveniles (Ayling et al. 1991). Such a pattern has been documented previously elsewhere on the GBR (Ayling et al. 1991), and cannibalism by coral trout (St John 1995) provides a potential mechanism for such an effect. A third alternative is that infringements of reef closures have erased any benefits from notional protection from fishing. Movement of coral trout among reefs, such that any benefits of reef closure would be diluted through migration of fish to and from areas where they can be caught (a fourth alternative), seems unlikely given recent work by Davies (1995a, b) which found very little evidence of inter-reef migration of *P. leopardus*. It is impossible to discriminate conclusively among these models without better reef-specific information about the amount of fishing effort and catch or levels of closure infringements, and without information about the relative status of reefs prior to the implementation of the previous zoning strategy.

Greater abundances of several other taxa on MNP reefs than on GU reefs are also difficult to explain. The fishes for which zone-related patterns were apparent were generally either a few chaetodon or pomacentrid species. At least some of these species are likely to be targets of the aquarium fish industry, and so might be expected to have suffered from collection on the GU reefs. Again, however, the dearth of information about the distribution and amount of such collection effort makes it impossible to correlate patterns in abundances with the (putative) cause of lower abundances. Although chaetodons have not been found to be major dietary items of target species for reef fisheries (e.g. coral trout), several pomacentrid species are commonly eaten by coral trout (St John 1995; Kingsford 1992). The consequence of reduced coral trout abundances on GU reefs (because of fishing), however, would be expected to be reduced predation pressure on such target species, which would presumably lead to increased abundances. Thus, we would have expected to see secondary effects of fishing (on GU reefs) manifest as increased abundances of prey species such as pomacentrids. Such effects were not apparent, however.

Perhaps of greater interest were the patterns in abundances of acroporid corals (both plates and others) and dead corals in relation to both zoning and tourist use. Acroporids were generally more abundant on non-tourist reefs than on tourist reefs, and more abundant on MNP reefs than on GU reefs. Dead corals, however, were more abundant on the more used reefs (tourist and/or GU) than on the putatively less-used reefs (non-tourist and MNP). It might be hypothesised that activities such as diving and anchoring might cause physical damage to corals and vessel discharges such as oil and waste might have adversely affected corals, both of which would be factors expected to vary with zoning history and use. As with other patterns related to use, however, there is little information about the relative intensities of use or about the cause-effect relationship between use and damage with which to substantiate or counter such an hypothesis (but see Roupheal and Inglis 1995). For example, whilst there is sound information about the presence of tourist pontoons on some reefs and the knowledge that those and other reefs are destinations of daily visits by large numbers of tourists, the level of use of many other reefs is poorly documented. Moreover, where moored structures provide a focus for tourist activity it might be expected that any impacts of those activities would be fairly localised. The effects that we found were based on the scale of entire reefs or habitats, substantially greater scales than the scales of activity of most site-specific tourism. There is no demonstrated mechanism by which such localised activities might affect benthos over greater scales, and even for the fishes, whose mobility might provide a vehicle for effects to be dispersed widely, previous research has found no demonstrable effects of high use sites on abundances (Sweatman 1996; Nelson and Mapstone 1998).

Although it cannot be inferred from this report that zoning history or notional intensity of tourist use have clearly been influential on the abundances of many reef biota, it would also be inappropriate to conclude that zoning was an ineffective or inappropriate conservation management strategy. In other arenas, area closure strategies have proved to be effective mechanisms for enhancing abundances of fished taxa locally, and relaxations of closures have clearly resulted in reductions in abundances of those taxa (Alcala 1988; Russ 1984, 1989, 1991; Russ and Alcala 1989). Although these few examples stem from environments in which the effects of local harvest were taxonomically more diverse, more destructive, and far more intense than is likely on the GBR, they do demonstrate that closure strategies can be effective, at least locally. There is a growing body of opinion, based largely on theory rather than empirical evidence, that area closure strategies may be the most productive conservation management strategies available for marine environments (Bohnsack 1994; Hilborn and Walters 1992; Plan Development Team 1990).

The patterns we describe, or lack of pattern, relative to the history of reef zoning also might reflect the effects of the initial choices of reefs to be included in each zone. Throughout the history of the GBRMP, the selection of reefs over which restrictions on use would be applied apparently has been heavily influenced by the public participation processes involved in the development of zoning plans. Combined with a shortage of information about the status of

reefs, and no clearly documented reef selection policy, these influences may have precipitated the selection of reefs for protection that were 'less productive' or 'less desirable' for activities such as fishing. The consequences of such confounding between past experience of use on reefs and future zoning status, if it occurred, cannot be known in the absence of good pre-zoning data.

Irrespective of the cause(s) of the patterns we recorded, the potential to infer effects of management strategies from simple one-off 'before and after' estimates of abundance or community structure seem poor. The considerable inter-annual variation in counts also suggest that simple paired surveys will be likely to produce large differences unrelated to effects of management strategy. As has been recommended repeatedly for assessments of environmental impacts, of which management might be seen as one, unequivocal inferences of effects of human activities will require repeated measurements over extended periods at both the impacted (managed) and control (used) locations (Keough and Mapstone 1995; Stewart-Oaten et al. 1986; Underwood 1991, 1993, 1995). Such a strategy does not currently exist on the GBR. Although it will now be possible to use data from the only long-term monitoring program under way (the AIMS Long-term Monitoring Program, Oliver et al. 1998) and targeted research such as the CRC Effects of Fishing Project (Mapstone, Campbell and Smith 1996) to infer effects of reef closure, such *ad hoc* inferences should not be seen as replacements for dedicated monitoring.

In this context, then, the most important message from this report may be that greater attention needs to be paid to the assessment and monitoring of management strategy than has been the case to date. In particular, we recommend that in developing amended zoning plans, the Great Barrier Reef Marine Park Authority explicitly consider the means by which the effectiveness of their management will be assessed and, as far as possible, accommodate the design of such assessments in the zoning plans. Facilitation of elegant designs for monitoring purposes is unlikely to be particularly influential in the allocation of areas to different uses, but should not be ignored. Of greater importance will be the explicit acknowledgment of the need for empirical assessments of management strategies, both in terms of socioeconomic and biological variables, the incorporation of such assessments into the 'learning' objectives of management planning, and the allocation of appropriate levels of funding for those assessments (Hilborn and Walters 1992; Ludwig et al. 1993; Walters 1986). There is increasing argument that explicitly constructing management strategy to maximise the potential for assessment and learning may be the fastest way to fine tune management to specific environments (Hilborn and Walters 1992; Ludwig et al. 1993; McAllister and Petermen 1992; Walters 1986; Walters and Hilborn 1976; Walters and Hilborn 1978; Walters and Holling 1990; Walters and Sainsbury 1990). The spatial extent and structure of the GBR, together with the relatively low use and stable political conditions in Australia relative to many other tropical countries, perhaps presents one of the greatest opportunities to implement such adaptive management strategies (Mapstone et al. 1996b). In the absence of such an explicit assessment and refinement approach, or formal and regular monitoring strategies targeted specifically at management strategy evaluation, most, perhaps all, interpretations of surveys of the type we have completed will be impaired by inadequate information and there will continue to be little empirical evidence from which to justify or refine existing management strategies.