

Review

The ecosystem effects of abalone fishing: a review

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Abstract. Although abalone fishing involves less damage to the habitat compared with other fishing methods, such as trawling and dredging and there are no problems of bycatch or discards, there are a number of issues relating to impact on the ecosystem. These issues include mechanical damage from anchors, catch bags and hoses, manipulation of predators and competitors by fishermen, and translocation of marine pests. The trophic impact of the removal of abalone would not be expected to be great, any impact may relate more to competition for space, for example, abalone may out-compete sea urchins for space when food is abundant. Therefore, the sea urchin population may be an ecological indicator of the impacts on the ecosystem of removing abalone. The data on the ecological impacts of abalone fishing are insufficient, and there have not been any direct experiments on the effects of varying abalone abundance on other ecosystem components. Recent studies on marine protected areas (MPAs) indicate that the ecological effects of fishing in reef systems may have had unexpected consequences. Abalone fishing offers a rare opportunity to trace the ecological effects of fishing, and to this end, a possible experimental framework is outlined.

Extra keywords: fishing effects, manipulative experiments, marine habitat, marine protected areas.

Introduction

Many fisheries around the world are in decline despite the best efforts of traditional target-species fisheries management. In this climate there is a growing awareness of the impacts of fisheries on the marine ecosystem and the need for ecosystem-based fisheries management (Jennings and Kaiser 1998; Hall 1999; Gislason *et al.* 2000). Fisheries are increasingly being required to show that they are ecologically sustainable. Ecosystem objectives should include the maintenance of ecosystem diversity, species diversity, genetic variability within species, directly impacted species, ecologically dependent species, and trophic level balance (Gislason *et al.* 2000). The direct effects of fishing can include mortality of target, bycatch and discard species, as well as damage to marine habitats from fishing gear. Fishing activities also lead to indirect effects on trophic and competitive interactions (Jennings and Kaiser 1998; Hall 1999; Gislason *et al.* 2000). Effective ecosystem-based fisheries management will require the identification of indicators with reference points that can be targets or limits (Trenkel and Rochet 2003).

In recent years, many abalone fisheries have collapsed (Hobday *et al.* 2001; Shepherd and Rodda 2001; Shepherd *et al.* 2001). In some cases the decline has been caused by overfishing whereas in others, pollution, natural recruitment failure and unknown causes have been implicated (Shepherd and Brown 1993). Like other fisheries, there is

a trend for ecosystem-based fisheries management to be applied to abalone fisheries. In Australia, for example, export approval for abalone is dependent on satisfying the criteria for ecological sustainability set up under Commonwealth legislation. As such, there has been an attempt to formalise the management of abalone fisheries within the framework of ecologically sustainable development. Research priorities include the development of robust environmental sustainability indicators, an increasing understanding of the interaction between abalone and other benthic competitors and predators (including exotics), and monitoring marine protected areas.

To understand ecological interactions and define sustainability indicators it should be possible to use experimental manipulations to investigate the effect of the abalone fishery on the coastal reef ecosystem. Results would allow us to quantify the effect of current fishing levels on the reef ecosystem, and assess possible changes to the ecosystem if fishing levels are altered in the future. Such experiments, however, need to be carefully planned so that the desired outcomes in terms of assessment of ecological impacts can be provided. Here I review the current body of knowledge on the relationship between abalone and the reef ecosystem, and the effects of abalone fishing on that ecosystem. Examples are drawn internationally, with a focus on the south-east Australian region – a region that produces almost two-thirds of the world's

reported wild abalone harvest (Gorfine and Dixon 2000). Recommendations are made for an experimental framework to determine the effects of abalone fishing on coastal reef ecosystems.

The ecosystem effects of removing the target species

Preamble

Any effects of reducing the abundance of abalone by fishing on the ecosystem will depend on the strength and direction of dependencies between abalone and other species. For example, the distribution and abundance of abalone on a reef might be heavily dependent on the presence of a second species but the reverse may not be true. Abalone interact with their ecosystem through several mechanisms, including feeding, competition, commensalism, predation and parasitism. The following section provides a review of studies relating to the interaction between abalone and other species. Where possible, the strength and direction of dependencies is assessed, and conclusions are drawn about the ecological effects of removing abalone from the ecosystem.

Feeding

Small, post-settlement abalone graze on benthic microflora, particularly diatoms, overlying crustose coralline algae, which are the preferred settlement substrate, although grazing on bacteria and directly on the coralline algae may also occur (Garland *et al.* 1985; Matthews and Cook 1995; Kawamura *et al.* 1998; Daume *et al.* 1999; Day and Branch 2002a). In the juvenile stage, at a size of approximately 5–10 mm, there is a transition to a diet of macroalgae (Kawamura *et al.* 1998, 2001). From this point the role of abalone in the ecosystem is more that of scavengers than grazers because they feed primarily on drift algae (Lowry and Pearse 1973; Shepherd 1973b). Abalone have a dietary preference for rhodophytes in Australasia and the tropics (Shepherd 1973b; Marsden and Williams 1996; Tahil and Juinio Menez 1999), while phaeophytes form the dominant diet in South Africa and the northern Pacific (Barkai and Griffiths 1986; McShane *et al.* 1994). Regional differences in diet preference may relate to factors such as the levels of phenolics in phaeophytes that inhibit feeding and the relative toughness of the algal thallus (Shepherd and Steinberg 1992; McShane *et al.* 1994; Stepto and Cook 1996). Higher levels of phenolics in phaeophytes of the Australasian region may relate to higher grazing levels (Steinberg *et al.* 1995). Seagrasses have also been found in the diet of *Haliotis laevigata*, *H. rubra* and *H. roei* in South Australia (Shepherd 1973b) and *H. fulgens* in Baja California Sur, Mexico (Serviere Zaragoza *et al.* 1998). Dietary analyses can also be influenced by bias due to differential digestion rates of different algal taxa (Foale and Day 1992; Day and Cook 1995). Overall, by feeding primarily on drift macroalgae, abalone appear not to have a structurally important role in the reef ecosystem in terms of feeding and diet (Shepherd and

Clarkson 2001). Notwithstanding this, the importance to the system of removing significant amounts of drift algae in areas of high abalone abundance is not understood, and the role of direct grazing on living seaweeds may be underestimated for some species (G. Edgar, personal communication).

Competition

Sea urchins are an important taxon with regard to community structure on many temperate reef systems where abalone are found. Fluctuations in the abundance of urchins in many areas leads to alternating ecosystem states between 'barrens' dominated by urchins and crustose coralline algae, and kelp dominated systems (Tegner and Dayton 2000). Sea urchins and abalone co-occur in reef ecosystems in the North Pacific, Australasia and South Africa (Tegner and Dayton 2000). Abalone and sea urchins are potential competitors for food because both feed on drift algae (Lowry and Pearse 1973; Shepherd 1973a; Andrew and Underwood 1992). However, sea urchins have a greater potential to modify reef ecosystems because they also graze directly on macroalgae (Andrew and Underwood 1992, 1993). Abalone tend not to occur in urchin barrens, implying that urchins are competitively dominant when food is limiting (Lowry and Pearse 1973; Andrew and Underwood 1992). When food is not limiting, however, abalone may be superior competitors for space (crevices; Lowry and Pearse 1973).

Several factors have been implicated in the fluctuations in sea urchin abundances, such as exploitation of predators (sea otters, lobsters, fish), recruitment, disease and physical disturbance (Tegner and Dayton 2000). The relative importance of competition with abalone is difficult to determine. This is particularly the case because, in general, fisheries for abalone pre-date those for urchins (Tegner and Dayton 2000), so that our perspective is coloured by historically lower abundances of abalone compared to urchins. In southern California, the apparent explosion in the sea urchin population in the 1950s and 1960s may have partially resulted from reduced competition with abalone (North and Pearse 1970).

The question in terms of the ecological effects of abalone fishing is whether correlations between abalone and urchin abundances only reflect the responses of abalone to urchin density, or whether the reverse can occur and urchins can be affected by abalone density. Most work would suggest that while abalone fluctuations may be affected by urchin abundances, the reverse is less likely. However, suggestions that abalone are superior competitors for space (Lowry and Pearse 1973) and that the decline of abalone may have led to increased urchin populations (North and Pearse 1970), indicate that abalone densities might influence urchin populations. Experimental work is required to test this hypothesis. If the hypothesis is correct, increased urchin abundance could be a potential indicator of the ecological effects of abalone fishing.

Commensalism

Remarkably, in some areas, including California, Japan and South Africa, urchins have a positive effect on abalone populations by providing shelter for juveniles under the spine canopy (Tegner and Dayton 2000). Juvenile abalone may benefit through protection from predators and possibly an enhanced food supply (Day and Branch 2002b). In South Africa, increases in the abundance of rock lobsters in some areas have resulted in a collapse in urchin numbers and a parallel decline in juvenile abalone (Mayfield and Branch 2000). It is not clear whether urchins are affected, positively or negatively, by this commensal relationship.

Another example of commensalism related to abalone is the community of organisms directly associated with the surface and underside of shells. One of these is the food-stealing limpet, *Hipponix conicus*, which favours the anterior margin of *Haliotis* shells for greatest access to food particles (Laws 1970). The potential impact of removing abalone depends on how specific the commensal organism's relationship is to the host. In the case of *H. conicus*, for example, hosts can include a range of gastropod genera other than *Haliotis*, so the impact on this species of removing abalone may be minor (Seddon 1993; Yamahira and Yano 2000).

Predation and parasitism

A variety of taxa prey on abalone, including: molluscs, such as whelks; octopus (Shepherd 1973b; Tegner and Butler 1985; Kojima 1988; Thomas and Day 1995); starfish (McShane and Smith 1986; Fujita and Seto 1998); crustaceans including crabs and rock lobsters (Shepherd 1973b; Day and Branch 2002b); fishes, such as wrasses (Shepherd 1973b; Shepherd and Clarkson 2001); and rays (Shepherd 1973b, 1990). It has been suggested that 11 arm starfish, *Coscinasterias muricata* may have a significant impact on *H. rubra* populations in localised areas of Port Phillip Bay, Australia (McShane and Smith 1986; Day *et al.* 1995).

The highly cryptic habit of juveniles is probably a response to their greater vulnerability to predation. Nocturnal feeding migrations of abalone may also be a response to avoidance of diurnal predators (Shepherd 1973b). Re-stocking experiments with hatchery-reared juveniles of red abalone, *H. rufescens*, suggested very high predation rates, particularly by octopuses (Tegner and Butler 1985). At localities in South Australia, wrasses are thought to be the dominant predator of abalone recruits and it has been suggested that these fish may control abalone recruitment (Shepherd 1998; Shepherd and Clarkson 2001).

The crucial question in terms of impact on the ecosystem of the fishery is the dependency of any predator on abalone prey. If abalone form only a small part of a broad diet then the reduction in abalone densities is unlikely to impact predator populations. Conversely, if predators tend to specialise in feeding on abalone then reduced abalone densities could

affect them. For example, although wrasses are a dominant predator on juvenile abalone in South Australia, they feed on a variety of invertebrates more or less in proportion with their abundances in the environment (Shepherd and Clarkson 2001). A strong response of wrasse populations to a decline in juvenile abalone density would therefore seem unlikely.

More studies are needed on other predator species to determine whether any strong dependence on abalone prey occurs. Anecdotal evidence suggests that juvenile abalone may be a major component of the diet of banded morwong, *Cheilodactylus spectabilis*, in south-eastern Australia (Gorfine and Dixon 2000), therefore, the diet of this species should be studied further with respect to abalone predation. The whelk *Haustrum baileyianum* (now *Thais*) may specialise on abalone. This whelk bores through abalone shells at the muscle attachment site and has not been found to prey on any other species (Thomas and Day 1995; R. Day, personal communication).

Parasites of abalone include shell-boring polychaetes, sponges and bivalves (Shepherd 1973b; Kojima and Imajima 1982; Oakes and Fields 1996; Del Carmen Alvarez Tinajero *et al.* 2001; Leonart *et al.* 2003). This form of parasitism may have serious deleterious effects on abalone growth and survival (Kojima and Imajima 1982; Oakes and Fields 1996; Leonart *et al.* 2003). Removing abalone from the ecosystem may have an impact on shell-boring polychaetes that infest abalone but these polychaetes are also found on shells of other mollusc species, and there is no evidence for a specific relationship with abalone (McDiarmid *et al.* 2004).

Background environmental variability

Interactions in the reef ecosystem exist against a background of high variability. In particular, climatic effects may cause major changes in reef systems. In the short-term, storms may have major effects on kelp cover and associated open spaces (Kennelly 1987; Tegner *et al.* 2001). In the longer term, climatic phenomena, such as the El Niño Southern Oscillation (ENSO) cycle and greenhouse warming may affect current patterns and water temperature that in turn influence recruitment patterns and survival (Tegner *et al.* 2001; Hobday and Tegner 2002). Another very important source of background variability is anthropogenic effects, such as pollution, sedimentation and habitat modification (Gislason *et al.* 2000; Hobday *et al.* 2001). The strength and direction of dependencies between abalone and other ecosystem components may be influenced by this background variability.

Direct effects of fishing practices

Abalone are collected by divers operating from boats and primarily using hookah gear, but also some scuba gear (Kailola *et al.* 1993). The abalone are prised from rocks using an 'abalone iron' (a rounded spatula-like tool) or screwdriver (Kailola *et al.* 1993). The boats are often small and launched

from trailers, allowing divers access to remote stretches of coastline. Abalone collected by the diver are placed in a catch bag that is sent to the surface when full. Collected abalone are sometimes kept alive in seawater tanks onboard the vessel. Recreational fisheries for abalone also occur in many areas, with abalone collected using snorkelling or scuba diving gear (Kailola *et al.* 1993).

Fishing techniques and practices can have significant effects on ecosystems. Practices, such as, trawling with nets and dredges and fishing with explosives have an obvious potential impact on the ecosystem. Compared to these, dive fisheries would intuitively be relatively benign. Further, problems of bycatch mortality or discard with other fishing methods do not exist in the abalone fishery. However, diving activities have been shown to have significant ecological effects, particularly in the tropics (Hawkins and Roberts 1992; Rouphael and Inglis 2001; Tratalos and Austin 2001). Divers damage corals and other organisms through direct physical contact with their hands, body equipment and, in particular, fins (Rouphael and Inglis 2001). Although damage done by individuals may be minor, cumulative effects may cause localised destruction of sensitive organisms (Rouphael and Inglis 2001). In a temperate example, recreational diving in a Spanish marine reserve had significant effects on colonial bryozoans, and recovery after the disturbance was slow (Garrabou *et al.* 1998). It has been suggested that disturbance through diving could lead to an alternative ecosystem state dominated by abrasion resistant species (Garrabou *et al.* 1998).

The ecological effects of diving practices used in abalone fishing have not been quantified. The dragging of the abalone catch bag has the potential to have impact on the substrate and algal canopy. Abalone fisheries use hookah breathing equipment in many countries and there may be an impact of dragging hookah hoses over the substrate. Moreover, there is anecdotal evidence for the practice of cutting back the algal canopy to minimise tangling of hookah hoses, and permit the passage of the tethered diver. Because the concentration of divers is a major factor in potential impact (Tratalos and Austin 2001), the effect of commercial abalone diving might be expected to be less severe than for recreational diving in localised areas. It would be possible to follow abalone divers and quantify any effects on the reef habitat (Rouphael and Inglis 2001), although diver behaviour may change under these circumstances (i.e. a more careful approach than normal is undertaken while under observation). In addition, controlled experiments could possibly be carried out to determine the effects of abalone fishing practices, using methods similar to those used in trampling studies (Povey and Keough 1991). Such experiments may suggest potential indicator organisms for damage through abalone diving practices, most likely sessile organisms with fragile skeletons and slow growth rates, particularly, erect, articulate and foliose species (Garrabou *et al.* 1998).

Direct effects on the ecosystem may also occur from other activities associated with abalone fishing. Anchoring has been found to have significant effects on coral reef and seagrass habitats (Davis 1977; Creed and Filho 1999; Milazzo *et al.* 2004), and has the potential to cause damage to temperate reef habitats. In some areas, abalone fishing is carried out while the vessel is drifting, so that anchoring damage is not be a problem. Another potential impact is when abalone are transported and act as a vector to translocate introduced organisms to uncolonised sites. An example of this occurred recently in central Victoria, Australia, where abalone with shells colonised by the introduced Japanese kelp, *Undaria*, were collected within Port Phillip Bay but the shells were discarded at the entrance to nearby Western Port Bay where the kelp had not colonised (G. Parry, personal communication). In this case the *Undaria* plants were removed before colonisation took place. This situation should not normally occur as part of normal commercial fishing operations because shucking abalone at sea is illegal in Victoria. However, translocation could potentially occur as part of normal operations, for example, if pest species are inadvertently transported on or in boat hulls. This translocation problem could be exacerbated where abalone are held in live tanks onboard fishing vessels and water is exchanged in a different area to where it is taken up. In this case, introduced phytoplankton, such as dinoflagellates, and planktonic cysts, spores or larvae of introduced species, could be translocated through water exchange.

Another direct effect of abalone fishing practices may be attempts by abalone divers to manipulate the ecosystem to improve abalone fishing. For example, there are anecdotal reports of divers removing seastars because of their perceived negative impact on abalone stocks; or crushing or destroying urchins or crabs in an effort to create space for abalone, or remove predators or competitors.

An exception to the conclusion of relatively moderate impacts of abalone fishing practices on the ecosystem occurs in some tropical areas. For example, in Indonesia, divers destroy significant areas of coral while collecting shellfish, including abalone, and lobsters ('reef gleaning'; Pet and Djohani 1998).

Ecological indicators

Ecologically sustainable management will require suitable indicators that can be measured against reference points that trigger management actions (Gislason *et al.* 2000; Rochet and Trenkel 2002). Three types of indicators have been proposed to assess the effects of fishing: (1) population based; (2) assemblage based (ignores interactions); (3) community based (i.e. trophic paths, biomass flows; Rochet and Trenkel 2002).

There is a strong research and conservation interest in biodiversity, and therefore, assemblage-based indices, such as, diversity and species richness may be widely considered.

However, species diversity has the problem of being composed of a mixture of species richness and evenness, with weightings that are not justified for most diversity indices. Most existing indices have little biological meaning, and their estimation is often impaired by technical difficulties (Hurlbert 1971). The simpler measure of species richness has the problem of being dependent on sample size and gear (Rochet and Trenkel 2002). A more detailed method of examining diversity is ordination or classification of assemblage data. Unfortunately, comparison of resulting classification plots are often subjective, and the development of objective methods of comparison is essential (Rochet and Trenkel 2002). However, the use of such multivariate methods may lead to the identification of indicator species that can be used as a proxy for assemblage level measures (Jennings and Reynolds 2000).

Given that the effects of fishing are likely to be subtler in a diver-based fishery compared to other fisheries, assemblage and community based indicators may not be sufficiently sensitive. The identification of so-called 'ecologically dependent' species may provide the most suitable indicators (Gislason *et al.* 2000). These species will show population responses to variation in abalone abundance caused by fishing. Examples would be predators that have a dependence on abalone prey or competitors that are sensitive to variation in abalone abundance. The indicator might be the abundance, growth rate, condition or some other trait of the dependent species, and reference levels would be based on these factors (Gislason *et al.* 2000).

An additional form of indicator that could be very useful in assessing impacts of abalone fishing would be habitat-based indices. Such indices could include percentage algal cover, area of bare rock and many other potential variables.

An experimental approach to assessing ecosystem effects of abalone diving

Two major problems usually confront the selection of indicator species for the effects of overfishing. One is that fishing has been undertaken for a considerable time and therefore pre-fishing conditions are unknown, making the determination of reference points problematic (Gislason *et al.* 2000). Second, correlations between fishing effects and ecosystem changes may be found but causation is very difficult to establish because the systems do not lend themselves to experimentation (Gislason *et al.* 2000). In the case of the abalone fishery, however, both of these problems can be dealt with. First, it is possible with subtidal reef ecosystems to use marine protected areas (MPAs) as a proxy for knowledge of pre-fishing conditions (Dayton *et al.* 2000; Gislason *et al.* 2000; Tegner and Dayton 2000). Second, unlike most other systems, communities occupying hard substrates, such as, subtidal reefs offer the opportunity to use experiments to determine dependence and causality (Hall 1999).

Manipulative experiments to assess the ecological impact of the abalone should simulate different levels of fishing, across a range of spatial and temporal scales. The first step in the process of establishing experiments would be to examine existing data for correlations between abalone abundances and other ecosystem components based on existing monitoring data. This analysis would provide an initial screening for species that may be affected by variation in abalone abundances and therefore should be measured explicitly in experiments. Because questions of ecosystem impact are framed around showing no effect of fishing, statistical power is a major consideration (Jennings and Kaiser 1998). Some assessment of the level of replication required for sufficient statistical power, based on pre-defined effect sizes of biological importance, could be made by analysing the variability in taxa collected in existing monitoring programs (e.g. Channel Islands Monitoring Program, California; Victorian Fisheries Abalone Monitoring Program, south-eastern Australia).

For manipulations, experimental plots would be set up to encompass more than one reef system along the coast to give greater generality to the results. If possible, these experimental locations would include MPAs, so that an inside/outside MPA treatment could be applied. This would allow comparison of results in fished and unfished environments to assess the affect of interactions with other fisheries and to help establish reference points for any indicators selected. In Victoria, south-eastern Australia, for example, the recent declaration of MPAs in coastal reef habitat would make this approach feasible. Temporary fishery closures would have to be negotiated for the non-MPA sites. Experimental plots would be set up where abundances of abalone would be manipulated. Levels of the removal treatment could be no removal (control), 150% increase, and 50% and 100% removal of visible abalone. The size of experimental plots would be a trade-off between having sufficient area for results to be meaningful at the reef scale, but not so large as to make manipulations logistically intractable. An area in the order of a quarter hectare could be trialed in the first instance. The 150% increase in treatment is based on the fact that present abalone biomass in southern Australia is thought to be approximately 40% of the pre-commercial fishing level (Anon. 2002). Sampling would be conducted seasonally and would include a transect to assess fish abundance and randomly placed quadrats to assess macroinvertebrate abundance and macroalgal cover. After sampling was conducted, removal treatments would be adjusted to the required levels (e.g. if abalone migrate into the complete removal treatment they will be removed). Industry cooperation would be sought in maintaining the abundance levels of abalone in plots. Sampling would need to be continued for at least 2 years. Results would indicate whether removal of abalone had any significant effect on coexisting species, and if so, allow the selection of indicator species and the determination of reference points.

Comparing experimental treatments within and outside MPAs would allow the potential for interaction with other fisheries to be assessed. During the 10-year period since declaration of marine reserves in Tasmania, the most striking change relative to reference sites was the increase in the abundance of large rock lobsters (Edgar and Barrett 1997, 1999; Barrett *et al.* 2003). Some fish species also showed increases in size and abundance (Edgar and Barrett 1999; Barrett *et al.* 2002). Although the mean size of abalone increased, there was a marked reduction in abundance in reserves due to a lack of small abalone relative to reference sites (Edgar and Barrett 1999; Barrett *et al.* 2003). There was also a decline in abundance of urchins but no significant change in algal communities (Barrett *et al.* 2003).

There are several explanations for the unexpected decline in abalone abundances. First, the decline may not be real but rather might reflect a change in the size of emergence from the cryptic juvenile habitat (Barrett *et al.* 2003). This change in behaviour could be due to the increasing presence of large abalone and/or predators, such as lobsters and wrasse (Barrett *et al.* 2003). Alternatively, the decline could be real, possibly resulting from increased competition with large abalone or increased levels of predation by lobsters, fish or octopus (Barrett *et al.* 2003). These results indicate that the interaction of multiple fisheries can result in complex effects on individual fishery species. Manipulative experiments will be required to determine whether declines in small abalone are due to behavioural factors or mortality (Barrett *et al.* 2003).

The implication from studies of MPAs in Tasmania is that rock lobsters may have a major influence on abalone behaviour or mortality (Barrett *et al.* 2003). Experiments could be extended to include manipulations of rock lobster abundance as well as abalone to test for interactions between these two groups. This would, however, add considerably to the logistical complexity of the experiments.

Conclusions

In general, a review of the literature relevant to the impact of abalone fishing on the ecosystem would suggest that the abalone fishery is relatively benign with respect to ecosystem impact compared with fishing activities, such as, trawling and dredging. There are also no problems of bycatch or discards as occurs in other fisheries. Despite this, there are some issues relating to ecological impacts that need to be considered. These issues include: mechanical damage to habitat from catch bags, hookah hoses and anchors; attempts by divers to reduce mortality of abalone by removing potential predators and competitors; and the potential for translocation of introduced species. Managers and the industry could tackle some of these issues through 'codes of practice' arrangements. In terms of the impact of removal of abalone the trophic impact would not be expected to be great. Abalone feed primarily on

drift algae and are themselves typically part of a generalised predator diet. In the latter case the reported high consumption of juvenile abalone by banded morwong may be worthy of further investigation. Any impact of abalone on the ecosystem may relate more to competition for space, for example, it is thought that abalone may out compete with urchins for space when food is abundant.

A caveat to conclusions of low ecosystem impact of abalone fishing is that the data in many areas are insufficient, and studies on ecosystem dependencies on abalone have largely taken place where abalone populations have already been reduced by fishing. Whereas experiments have been carried out on the influence of other taxa on abalone (Andrew *et al.* 1998), no specific experiments have been carried out on the impact of abalone removal on the reef ecosystem. An intuitive conviction that abalone fishing has minimal ecosystem impact should not circumvent rigorous experiments, particularly when abalone offer one of the few fishing situations where experiments would be tractable. The results from 10 years of data in Tasmanian MPAs show that ecosystem responses can be quite unexpected, with a reduction in abalone abundance but an increase in size (Barrett *et al.* 2003). The suggested interaction between rock lobsters and abalone in this observation is worthy of experimental investigation.

A prerequisite to conducting experiments on ecosystem impacts of abalone fishing would be analysis of existing data sets on ecosystem components collected as part of monitoring surveys. Correlation analysis would indicate taxa that could be potentially influenced by changing abalone abundance, and variability in these taxa would suggest the necessary level of replication required for sufficient statistical power. It is possible that for some taxa the required replication levels would not be logistically feasible.

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