

Establishment of a No-take Area (NTA) could not Guarantee the Preservation of Coral Communities in Chinwan Inner Bay, Penghu, Taiwan

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Hernyi J. Hsieh, Kao-Song Chen, Yihan I. Lin, Yuanchao A. Huang, Andrew H. Baird, Wan-Sen Tsai, Wei-Cheng Su, and Chaolun A. Chen (2011) Establishment of a no-take area (NTA) could not guarantee the preservation of coral communities in Chinwan Inner Bay, Penghu, Taiwan. *Zoological Studies* 50(4): 443-453. No-take areas (NTAs) are an increasingly prevalent approach to conserve coral reefs, but are not always a guarantee of preserving pristine conditions if the area lacks ecological resilience, proper management, or both. In this study, the NTA of a subtropical coral community at Chinwan Inner Bay (CIB), Penghu, Taiwan, established in 2005, showed a 5-fold decline in living coral cover from 80.9% ± 10.8% in 2001 to 16.3% ± 8.4% by 2008 with the dominant coral functional group shifting from branching species to columnar and massive species. Analyses of fish functional groups and macroinvertebrates showed strong signs of overharvesting. In addition, an outbreak of and nearly 4-fold increase in the corallivorous gastropod, *Drupella* spp., from 5.5 individuals (ind.)/m² in 2003 to 22.7 ind./m² in 2009, was believed to be responsible for the disappearance of major coral functional groups at CIB. Depletion of fish and invertebrate functional groups, the outbreak of a corallivorous gastropod, and previously existing and continuing anthropogenic-induced degradation of the marine environment suggest that the overall loss of ecological resilience provided little chance for the coral community to bounce back to the pristine status described in 2001, even after the CIB NTA was formally established in 2005. <http://zoolstud.sinica.edu.tw/Journals/50.4/443.pdf>

Key words: Chinwan Inner Bay, No-take area, Functional group, Coral community, Outbreak of *Drupella*.

Coral reef ecosystems have declined in health due to a long history of overfishing, pollution, habitat destruction, and synergetic effects due to impacts of climate change (reviewed in Hoegh-Guldberg et al. 2007). No-take areas (NTAs), where fishing and other human activities are prohibited, are an increasingly prevalent approach to prevent coral reefs from continuing degradation, and it is hoped that they can “buy time” for coral reefs to acclimatize to future climate changes (Hoegh-Guldberg et al. 2007).

The success of NTAs in coral reef management relies on 2 main factors: the capacity of ecological resilience and the efficiency of governances and law enforcement of the NTA (Bellwood et al. 2004, Mora et al. 2006). The persistence of diverse critical functional groups, both herbivores and corals, in coral reefs ensures the potential resilience of local ecosystems from disturbances (Bellwood et al. 2004). Recent studies on coral reefs in the Bahamas demonstrated that reduced fishing pressure and weak predator-prey

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interactions within marine reserves can create trophic cascades, which increase the number of grazing fish, reduce the coverage of macroalgae, facilitate coral recruitment, and thus, enhance the recovery of coral (Mumby et al. 2006 2007, Mumby and Harborne 2010).

The efficiency of governance and enforcement of laws by management in NTAs are critical parts of the balance between conservation and sustainable utilization of resources in coral reefs (Sale 2002 2008, Sale et al. 2005, Mora et al. 2006). Even though 18.1% of coral reefs are located in the marine protected areas (MPAs) worldwide, only 1.6% of reef areas are within adequately managed MPAs, and < 0.1% of the world coral reefs are within NTAs with no poaching (Mora et al. 2006). It was suggested that the global network of MPAs is very inefficient in protecting coral reefs. The factors contributing to this failure include regulations on extraction,

poaching, external risks, and the size and isolation of the MPAs (Mora et al. 2006). On the contrary, socioeconomic and ecological analyses of coral reef conservation efforts at 4 national parks, 4 co-managed reserves, and 3 traditionally managed areas in Indonesia and Papua New Guinea demonstrated that MPAs are not the answer, and that in fact, support of local communities is far more important than some government-mandated fishing closure (i.e., NTAs) (McClanahan et al. 2006).

In the present study, declines in living coral cover, functional fish, coral, and macroinvertebrate groups, and an outbreak of a corallivorous snail, *Drupella* sp., are described in the NTA of Chinwan Inner Bay (CIB), Penghu, Taiwan (Fig. 1). The pristine coral community at CIB was described as having healthy and high living coral cover (LCC) of up to 80%, and was first proposed to be an MPA in 2001 (Figs. 2A, B, Hsieh et al. 2001). CIB was

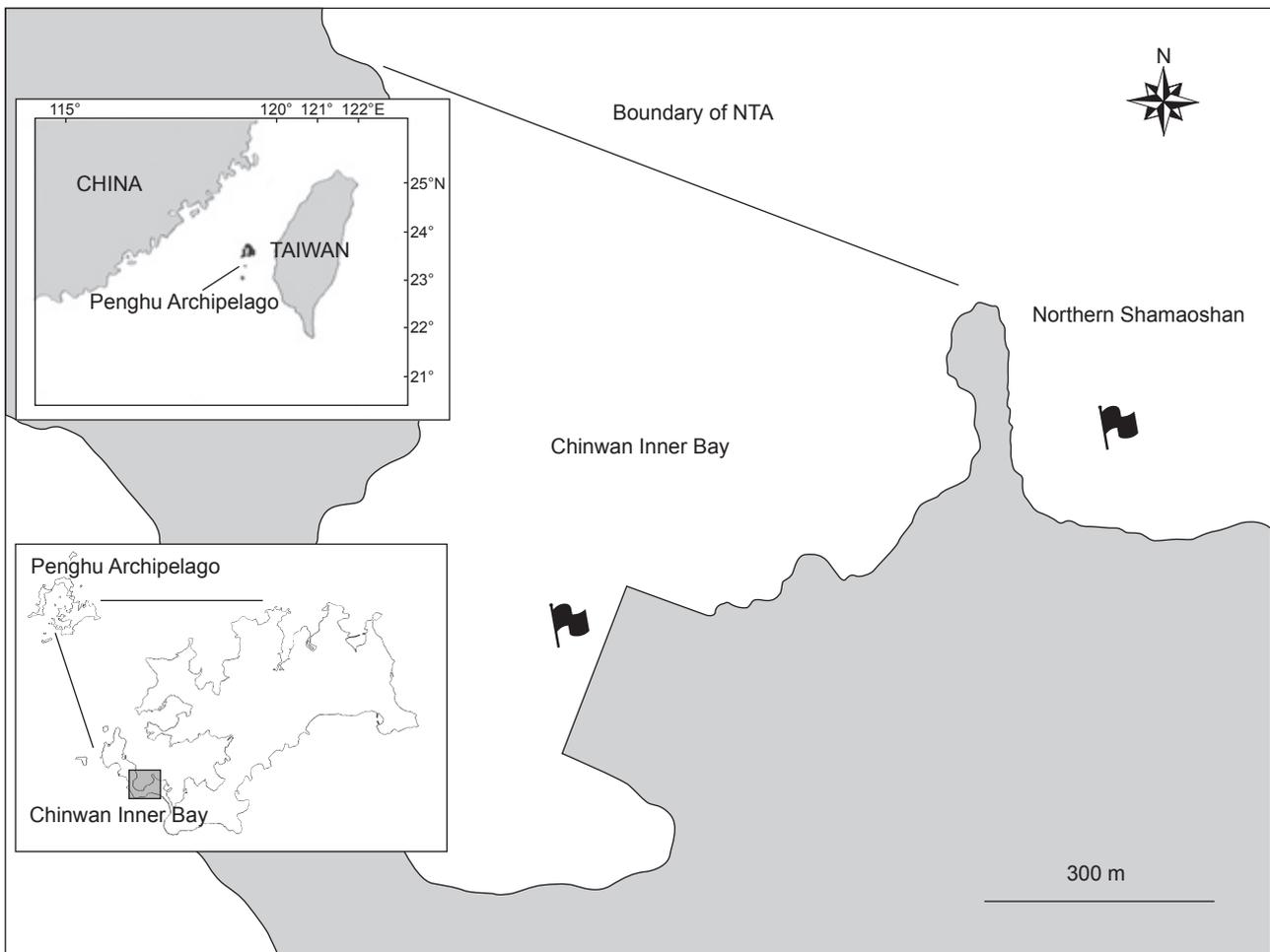


Fig. 1. Map of Chinwan Inner Bay and the design of the no-take area modified from Hsieh et al. (2007). Study sites are denoted as flags.

declared the 1st NTA in Penghu by legislation by the local county township in 2005. Socioeconomic analyses showed that most of the fishing at CIB is for recreational purposes characterized by low investment, low harvest rates, high dependence on weather conditions, and self-consumption. It was suggested that no direct conflict of interests with local communities at 2 neighboring villages and conservation support could guarantee the success of the CIB NTA (Hsieh et al. 2007). However, a lack of ecological resilience suggests that the NTA at CIB failed, and the coral community in the area has little chance of bouncing back to the pristine status described in 2001.

MATERIALS AND METHODS

Study site

CIB (23°31.804'N, 119°33.600'E) is a semi-enclosed bay located in an ancient crater to the south of the inner sea at Penghu, Taiwan in the

Taiwan Strait (Fig. 1). Detailed geographic and topological characteristics of CIB were described by Hsieh et al. (2007). LCC in CIB was high and dominated by branching species, such as *Acropora muricata* and *Montipora cactus* (Hsieh et al. 2001, Hsieh 2008).

Monitoring coral communities, fish, macroinvertebrates, and an outbreak of *Drupella* spp. at CIB

Coral communities were monitored in shallow (1-3 m) and deep (6-8 m) waters of CIB in 2001, 2003, and 2008 using a video transect method described by English et al. (1997). Six 20-m transects were laid out in each zone. Benthic organisms underneath the transects were recorded with a digital video recorder (Sony TRV-950) with a wide-angle lens at a distance of 30 cm to capture the maximum area and resolutions. The 20-m transect can be treated as a continuous belt consisting of 80 25 × 25-cm frames. A random number generator (Excel, Redmond, Washington,

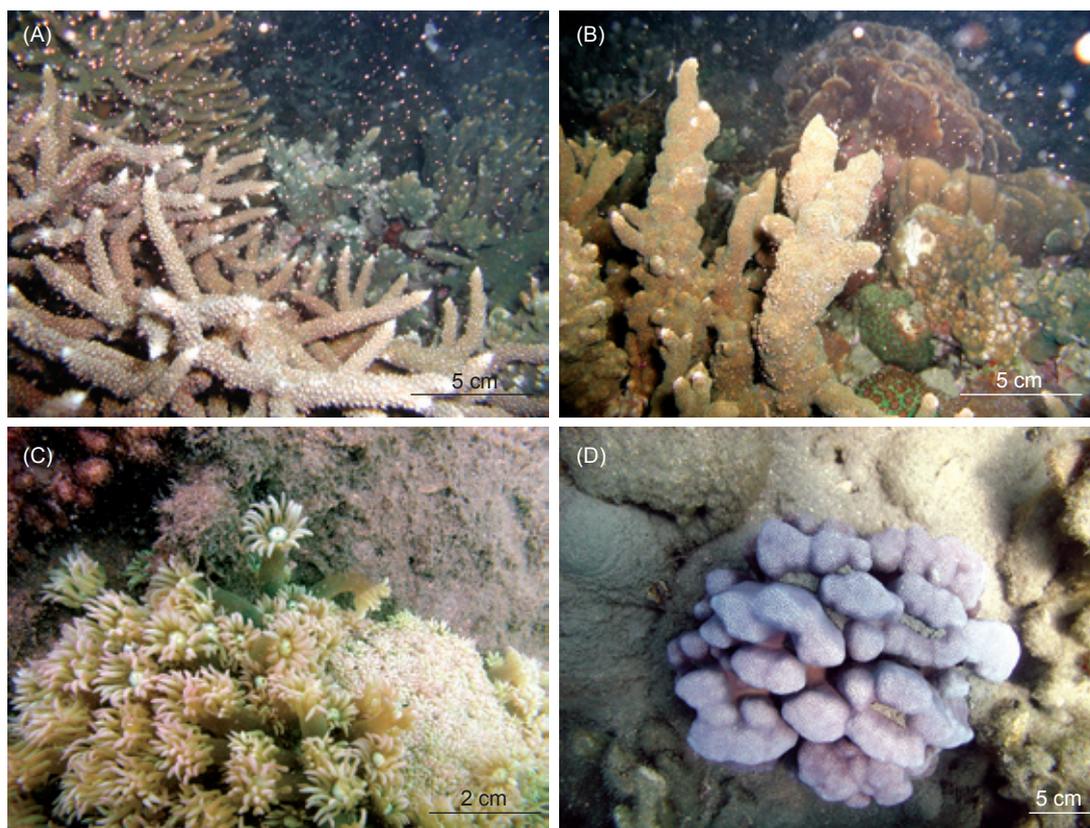


Fig. 2. Example of coral species in Chinwan Inner Bay. Colonies of dominant branching species which took part in the mass spawning in 2001 included (A) branching *Acropora muricata* and (B) *Montipora cactus*; the degraded community covered only by massive coral species in 2009 (C) *Gonipora columna*, and (D) deformed *Porites lutea* on the top of dead branching coral rubble or sediments.

USA) gave us non-overlapping information from which to choose the target frames. In total, 24 frames, each of which was 25 × 25 cm, were randomly chosen from the footage of each transect by the software, iMovie (Rubin 2004). Images of each frame were processed using Adobe Photoshop (San Jose, California, USA), and transferred to a Coral Point Count with Excel extension (CPCe) for point counts (Kohler and Gill 2006). Thirty random points were automatically spread onto frames, and the codes corresponding to the recognized objects in the images were denoted. Scleractinian corals were identified to species based on a field identification guide established in pilot studies (and also by referring to Nishihara and Veron 1995, Veron 2000, Wallace 1999) and others to genera if possible. Live corals were identified by recognizing normally colored colonies with no bleached or dead skeletons or with filament or turf algae. Live coral cover was defined as the relative portion of point counts of live corals divided by the total number of random points checked.

Indicator species of fish and macroinvertebrates were surveyed in 2001, 2003, and 2006 using rapid assessment techniques and quantitative surveys modified from ReefCheck methodology (Hodgson 2000, Table 1). A 100-m transect was laid out on the top of the coral community parallel to the shoreline. Divers examining fish and macroinvertebrate species swam along the transect while visually counting numbers of indicator species listed on the ReefCheck underwater ID card, with every 20 m as

1 section, and with a 5-10-m interval between the next 20-m section. Fish and macroinvertebrates recorded were combined from 4 sections along each transect due to the extremely low abundance of each target species (see "RESULTS").

Corallivorous snails, *Drupella* spp., were noted in 2001 during the 1st survey in CIB (Hsieh unpubl. data). In 2003 and 2009, population densities and prey preferences of *Drupella* were surveyed using transect-line and random-quadrat methods. For comparison, the same sampling methods were applied to the coral community in the northern part of Shamaoshan (nSMS), which is outside CIB. One transect line (30 m long) was set up seaward and perpendicular to the shoreline until no snails or coral colonies were found. Three 1-m² quadrates were laid out at every 10-m interval. All snails encountered were collected and counted. When the feeding behavior was observed, *Drupella* snails were commonly found on feeding scars. In addition, they tended to hide beneath the dead skeletons (rubble) of branching corals and even buried themselves in the sand. The positions (scars of corals, dead corals, and sand) where *Drupella* snails were found and the host coral species were also recorded in 2003. Student's *t*-test was used to test differences in the mean densities of *Drupella* between 2003 and 2009. A Chi-square test was used to examine host and habitat preferences of *Drupella* between CIB and nSMS. All statistics were performed using the software PAST (Palaeontological Statistics; <http://folk.uio.no/ohammer/past>) (Hammer et al. 2001).

Table 1. Population densities (individuals/ 50 m × 5 m) of major fish and invertebrate functional groups in Chinwan Inner Bay (CIB) in 2001, 2003, and 2006

Occurrence	2001		2003		2006	
	3 m	6 m	3 m	6 m	3 m	6 m
Depth						
Fish						
Butterflyfish	48	12	20	53	46	55
Grouper	8	5	1	3	2	1
Humpback grouper	1	2	0	1	0	1
Parrotfish	0	1	1	0	0	0
Invertebrates						
Sea urchin	1	1	0	0	0	0
Edible sea cucumber	0	0	0	0	0	0
Lobster	0	0	0	0	0	0
Giant clam	0	0	0	0	0	0
Triton snail	0	0	0	0	0	0
Crown-of-thorns starfish	0	0	0	0	0	0

RESULTS AND DISCUSSION

The 8 yr of monitoring of the coral community in CIB, Penghu, Taiwan showed a trend of continuous decline, even though an NTA was formally established after legislation was passed in 2005 (Hsieh et al. 2007). The characteristics of the depletion of the scleractinian coral community at CIB included: (1) a strategy shift in coral morphological functional groups; (2) overfishing of major functional groups of fish and macroinvertebrates; (3) an outbreak of the corallivorous snail, *Drupella* sp.; and (4) eutrophication due to nearby marine cage culture (Huang et al. 2010) that resulted in loss of ecological resilience of the coral community in CIB.

“Strategy shift” in coral functional groups

LCC in CIB was relatively high with an average of $80.9\% \pm 10.8\%$ (Hsieh et al. 2001) of the benthos represented by scleractinian corals in 2001 (Figs. 2A, B, 3). Dramatic changes in the dominant scleractinian species were found among the years 2001, 2003, and 2008 (Table 2). Dominant scleractinian species in 2001 were *Galaxea fascicularis* (24.54%), *Montipora cactus* (16.68%), *Lithophyllon undulatum* (8.74%), and *Acropora muricata* (6.74%). In addition, a rare coral species, *Euphyllia paraancora*, was found at 6 m deep with an average coverage of 6.26% (Hsieh et al. 2001). These 5 species represent 3 major morphological functional groups, branching (*A. muricata* and *M. cactus*), massive (*G. fascicularis* and *E. paraancora*), and encrusting (*L. undulatum*), suggesting that the coral community at CIB deserves the highest priority of conservation (Hsieh et al. 2007), based on its value classified by coral morphology (Edinger and Risk 2000). In 2003, significant decreases in the LCC in 5 of the

6 major coral species in the CIB were recorded. *Galaxea fascicularis* dropped to 5.16%, *M. cactus* to 3.74%, *L. undulatum* to 0.62%, and *E. paraancora* had completely disappeared (Fig. 3). Although the relative LCC of *A. muricata* increased from 6.74% to 13.30% due to the decreases in other coral species, the trend of a decline in the LCC was not reduced even though an NTA was established in 2005 (Hsieh et al. 2007). A video transect survey conducted in 2008 showed that degradation of the coral community in CIB not only involved a decline in the total LCC to 16.27%, but also coral species shifted from previous dominant groups to either columnar or massive species, such as *Goniopora columna* (4.00%) and *Porites lutea* (3.84%) (Figs. 2C, D). *Acropora muricata*, *M. cactus*, and *E. orpheensis* were not recorded in the survey in 2008 (Fig. 3), suggesting that the coral community at CIB had undergone a dramatic “strategic shift” (Done 1999) from pristine to a stressed status in 8 yr.

A phase shift describes the change in state from corals to alternative communities (most notably macroalgae) (Hughes 1994, Hughes et al. 2010). Long-term ecological studies on a Bermuda reef showed that herbivorous fish could reduce the coverage of macroalgae, facilitate coral recruitment, and thus, enhance the recovery of corals (Mumby et al. 2006 2007, Mumby and Harborne 2010). A strategy shift depicts a change taking place between contrasting coral states that contribute in different ways to reef growth (Done 1999). The classical case of a strategy shift is the very large massive and perennial *Porites* corals killed by crown-of-thorns starfish in the Indo-Pacific and quickly replaced by rapidly growing and ephemeral table *Acropora* within a decade or so (reviewed in Done 1999). The Caribbean-wide, synchronous mortality among large staghorn coral, *A. cervicornis*, caused by different band

Table 2. Live coral cover (LCC) and relative abundances (RAs) of individual dominant species in 2001, 2003, and 2008

2001	RA (%)	2003	RA (%)	2008	RA (%)
<i>Galaxea fascicularis</i>	24.54%	<i>Acropora muricata</i>	13.30%	<i>Goniopora columna</i>	4.00%
<i>Montipora cactus</i>	16.68%	<i>Galaxea fascicularis</i>	5.16%	<i>Porites lutea</i>	3.84%
<i>Lithophyllon undulatum</i>	8.74%	<i>Montipora cactus</i>	3.74%	<i>Goniopora djiboutiensis</i>	2.04%
<i>Acropora muricata</i>	6.72%	<i>Echinophyllia orpheensis</i>	3.12%	<i>Pavona decussata</i>	2.01%
<i>Euphyllia paraancora</i>	6.26%	<i>Pavona frondifera</i>	1.54%	<i>Pavona frondifera</i>	1.50%
<i>Echinophyllia orpheensis</i>	5.22%	<i>Platygyra sinensis</i>	1.54%	<i>Turbinaria mesenterina</i>	0.88%
LCC	80.82%	LCC	34.06%	LCC	16.27%

diseases (reviewed in Rosenberg and Loya 2004) and several hurricanes (McField 2000, McField et al. 2001) resulted in opportunistic coral, such as *Agaricia* spp., to overgrow on the stacks of dead rubble formed by *A. cervicornis* in Belize (Aronson and Precht 1997 2001). In both cases, no overgrowth of macroalgae on the open substrates after a disturbance was observed, suggesting no phase shift was seen in the catastrophic change of coral communities to alga-dominated reefs in the Caribbean (Hughes 1994, Gardner et al. 2003). In CIB, the reduction in LCC was accompanied by a loss of branching functional groups and a shift to dominance to either columnar or massive groups which are thought to be more stress-tolerant (Edinger and Risk 2000). The dead *Acropora* and *Montipora* rubble remained as open substrates, and no macroalgal blooms occurred during the 8-yr study period (Hsieh et al. 2008), supporting the concept that change in the coral community in CIB was under a process of a strategy shift.

Overfishing of the major functional groups of fish and macroinvertebrates

Dai (1997) mentioned that overfishing was one of the major factors which resulted in coral habitat degradation in Taiwan. Yearbooks of the Fishery Agency (<http://www.fa.gov.tw/>) since 1951 provide detailed information about fishing vessels

and annual catches in Penghu. Time-series plots show an obvious shrinking trend of catch/tonnage, which suggests a possible overfishing scenario. This clearly indicates that diversified fish groups ensure the ecological function of coral reef ecosystems, and the overharvesting of different fish functional groups resulted in the local or regional extinction of more than 50 fish populations in the last century (reviewed in Bellwood et al. 2004). Although coral functional groups shifted from diversified growth forms of acroporid species to stress-tolerant species of massive corals, and no phase shift was observed at CIB, severe overfishing of major functional groups, as a consequence, might have pushed the coral community of CIB to a level where it was unlikely to be ecologically resilient. CIB has been a fishing ground for local villages for centuries (Hsieh et al. 2007). No local community control or legally enforced regulations were imposed to manage the fish or invertebrate populations, until the CIB NTA zone project was implemented in 2005. Hsieh et al. (2007) suggested that there was less direct conflict of interests with local communities at 2 neighboring villages, and conservation support could guarantee the success of the CIB NTA based on coral community conservation. In this study, surveys in 2003-2008 all showed extremely low or no large predators (groupers) or herbivorous fish (parrotfish), or large

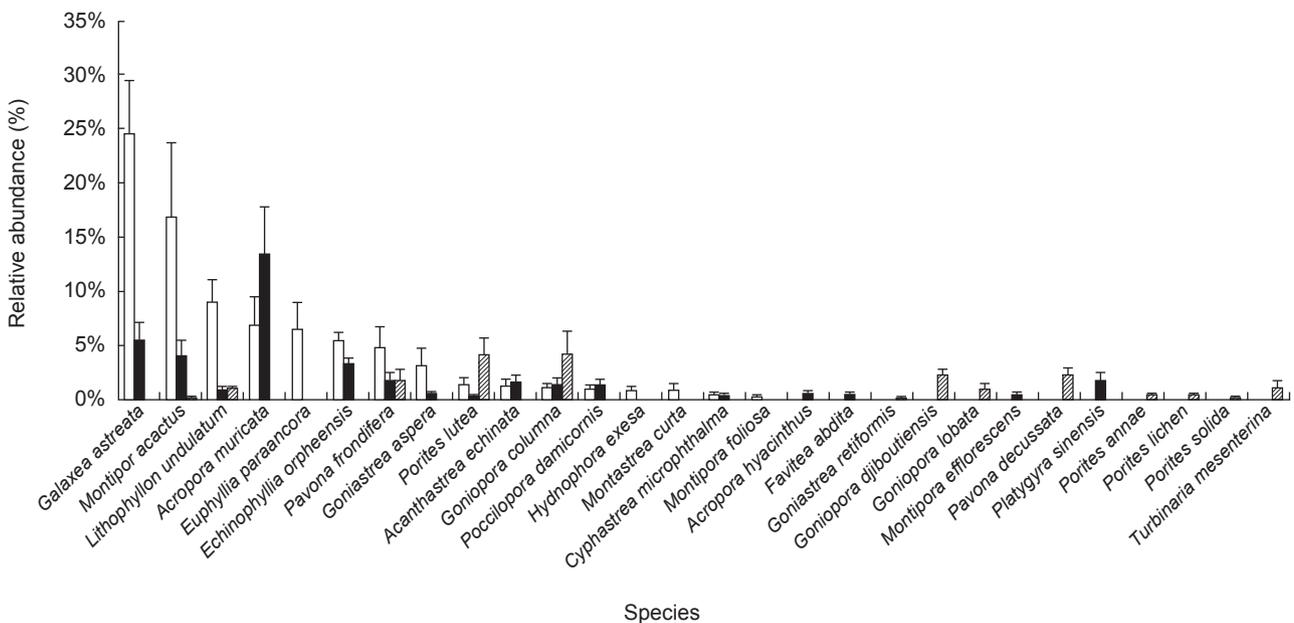


Fig. 3. Relative abundances (mean \pm S.D., $n = 6$) of dominant species which comprised the coral community at Chinwan Inner Bay in 2001 (\square), 2003 (\blacksquare), and 2008 (///).

macroinvertebrates (sea urchins, lobsters, giant clams, etc.). Top-down alteration of food webs by removing large predators and herbivorous fish was clearly observed at CIB (Table 1). We suggested that depletion of the major fish and invertebrate functional groups probably occurred long before this study commenced, even though corals looked healthy and the LCC was still relatively high in 2001 (Fig. 1, Hsieh et al. 2001).

The health of a reef is also greatly influenced by the abundance and diversity of marine organisms. They are a remarkably diverse group of animals due to the variety of functions they perform such as nutrient recycling, the removal of detritus material, and the consumption of invading algae (Hutchings 2003). Disturbances to marine invertebrates can result in negative ecological impacts upon reefs, such as algal blooms and invertebrate plagues leading to reductions in coral cover (Hutchings 1986, Dumas et al. 2007). A similar pattern of overharvesting observed in fishes was also seen in the even-worse depletion of macroinvertebrates in CIB such that almost none of the target invertebrate species was observed

during the 8 yr of surveys. This was probably due to the intensive collection of these invertebrates for food by local villages on the intertidal reef flat during low tide in CIB (Hsieh et al. 2007).

Outbreak of the corallivorous snail, *Drupella* spp.

In 2003, the corallivorous snails, *Drupella* spp. (Fig. 4) was mainly found on colonies of *G. fascicularis* (48.15%), *M. cactus* (37.04%), *A. muricata* (7.41%), *E. orpheensis* (3.7%), and *L. undulatum* (3.7%) at CIB. In the nSMS, *Drupella* spp. were mainly found on Acroporidae coral, with the highest frequency on *M. cactus* (58.33%) and < 20% on *A. muricata*, *A. humilis*, and *A. valida*. Prey coral preference significantly differed between CIB and nSMS (χ^2 test = 14.975, $p < 0.05$, Table 3). The number of *Drupella* found on scars of coral colonies was significantly higher at CIB than at nSMS (χ^2 test = 44.296, $p < 0.0001$, Table 4) with an average density of *Drupella* individuals at CIB (5.5 ind./m²) over 3-fold higher than that at nSMS (1.72 ind./m²) (Table 4). The *Drupella* density at CIB in 2003 had reached the criteria of an outbreak



Fig. 4. Massive and encrusting species immune to the grazing of *Drupella* spp. were sparsely distributed on the rubble of the former branching coral in 2009.

according to estimations by field surveys (Cumming 2009). In 2009, a 2nd survey showed an over 4-fold increase in the density reaching 22.7 ind./m² *Drupella* at CIB.

A variety of marine invertebrates feed, either facultatively or obligately, on the living tissues of scleractinian corals. Once an outbreak of feeders occurs on a reef, depletion of large areas of coral may be observed. The best-known case is the crown-of-thorns (CoTs) starfish, *Acanthaster planci*, that was reported to cause significant destruction of coral reefs worldwide. Impacts of CoTs on the Great Barrier Reef (GBR) have been felt for nearly 50 yr. At peak densities, the starfish killed substantial portions of reef corals in localized areas of the tropics. But its impact on reef ecosystems in recent years seems to have had detrimental effects on already stressed coral communities (e.g., due to coral bleaching) and reduced the abundance of corals to very low levels (Sweatman et al. 2008).

Similar to outbreaks of CoTs, the corallivorous gastropods *Drupella* spp. are best known as

gastropod agents of large-scale disturbance to coral reefs, particularly in Western Australia and Japan, where population outbreaks drastically reduced coral cover (reviewed in Cumming 2009). High-density populations of *Drupella* spp. were implicated in reef damage in Western Australia (Stoddart 1989, Turner 1992 1994), the Red Sea (Al-Moghrabi 1997, Shafir and Rinkevich 2008), Kenya (McClanahan 1994), Japan (Kimura et al. 2005), the Philippines (Moyer et al. 1982), the Marshall Is. (Boucher 1986), and Hong Kong (Cumming and McCorry 1998). It is now generally accepted that the intensity, extent, and features of the damage caused by *Drupella* spp. are similar to those resulting from *Acanthaster planci*.

Cumming (2009) reviewed the current knowledge of *Drupella* outbreaks around the world and concluded that evidence exists for population outbreaks of *Drupella* in only 3 broad locations: Japan, the northern Red Sea, and Ningaloo Reef in Western Australia. These are the only locations where *Drupella* densities of > 3 ind./m² were recorded. All 3 locations include some sites with

Table 3. Frequency distribution of preferred coral prey of *Drupella* snails in Chinwan Inner Bay (CIB) and north Shamaoshan (nSMS) in 2003

Coral species	CIB* No. of predated individuals	nSMS* No. of predated individuals
<i>Acropora humilis</i>	0	2
<i>A. valida</i>	0	1
<i>A. muricata</i>	2	2
<i>Echinophyllia orpheensis</i>	1	0
<i>Galaxea fascicularis</i>	13	0
<i>Lithophytum undulatum</i>	1	0
<i>Montipora cactus</i>	10	7
Total no. of predated individuals	27	12

*Chi-square test showed a significant difference of *Drupella* preference of coral species between the 2 sites (χ^2 -test = 14.975, $p < 0.05$).

Table 4. Frequency distribution of the occurrence of *Drupella* snails and populations densities in Chinwan Inner Bay (CIB) and north Shamaoshan (nSMS) in 2003

Occurrence	CIB*	nSMS*
Scars of live corals	163	31
Dead corals	35	21
Sandy bottom	0	10
Total no. of predated individuals	198	62
Population density (no./m ²)	5.50	1.72

*Chi-square test showed a significant difference of *Drupella* preference on coral species between the 2 sites (χ^2 -test = 44.296, $p < 0.0001$).

> 5 ind./m². In comparison to outbreak densities of CoTs, Cumming (2009) suggested that densities which distinguish non-outbreak from outbreak populations of *Drupella* are 1.4-6.4 ind./m². The density of *Drupella* at CIB in 2003 (5.5 ind./m²) fit in this range of an outbreak compared to the non-outbreak density at nSMS in the same year (1.72 ind./m²). These data also imply that the *Drupella* outbreak in CIB might have begun well before the commencement of the study in 2001. The *Drupella* density of 22.7 ind./m² in 2009 is the highest record compared to outbreaks in Japan, Western Australia, and the Red Sea (Cumming 2009).

The causes underlying *Drupella* outbreaks are not well understood. Several hypotheses were proposed including (1) outbreaks are the result of natural fluctuations in the natural population size of *Drupella*, (2) stochastic processes determine recruitment success, (3) anthropogenic influences reduce the mortality of *Drupella* because of overfishing of snail predators (e.g., the fish *Coris aygula*), and (4) modifications of coastal environments result in increased runoff and siltation (reviewed in Turner 1994, Carpenter 1997). The former 2 related to life history traits remain to be further investigated for the *Drupella* population at CIB. The latter 2; however, might account at least in part for the *Drupella* outbreak, and subsequently led to a failure of resilience of the coral community at CIB, even though an NTA was established in 2005 (Hsieh et al. 2007).

Environmental degradation and loss of ecological resilience

The loss of different functional groups at CIB, including depletion of large predators, herbivorous fish, and macroinvertebrates, and the outbreak of a corallivorous gastropod, might have led to a crash of ecological service cascades for maintaining a healthy coral community. On the other hand, environmental degradation, including eutrophication caused by marine cage culture (Huang et al. 2010), sewage runoff from adjacent townships, and unregulated intertidal collection and poaching (Hsieh unpubl. data) might also have produced long-term impacts from “bottom-up” effects which accelerated degradation of the coral community at CIB.

Scientific evidence, including ecological, fishery, and socioeconomic analyses, provide important criteria to evaluate the conservation value of local and regional coral reefs and to

establish, monitor, and evaluate MPAs from as broad a range as the Chagos in the Indian Ocean (Koldewey et al. 2010) and the northwestern Hawaii Is. (Kittinger et al. 2011) to small marine reserves like the NTA at CIB. Nevertheless, fundamental and relatively healthy coral and its associated community are biological and ecological keystones to the success of an MPA. It is suggested, based on scientific evidence, that the coral community at Chinwan Inner Bay has high conservational value (Hsieh et al. 2007). Although the very 1st NTA in Penghu was established by enforcing local legislation in 2005, loss of ecological resilience through depletion of diverse functional groups, *Drupella* outbreaks, and anthropogenic-induced environmental degradation demonstrate that there is no guarantee that an NTA can preserve the pristine coral assemblages at Chinwan Inner Bay, Penghu, Taiwan.

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