STATUS AND CONSERVATION OF THE REEF GASTROPOD *Trochus niloticus* IN THE PHILIPPINES

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For the unsung heroes in the middle of the Sulu Sea - the Rangers of Tubbataha Reefs Natural Park;

And for the marginal Filipino fishermen who are so dependent from the sea's bounty.

ABSTRACT

The most commercially important reef gastropod *Trochus niloticus* mainly harvested for the production of mother-of-pearl buttons, is highly susceptible to over-exploitation. Although conservation measures began shortly after the start of its commercial harvest in the early 1900s, their populations have been severely depleted prompting some countries like the Philippines to declare it as a threatened species. With the country's limited success in conserving trochus, this thesis explores the status of trochus in the wild, on-going conservation measures and some aspects of its biology in Palawan, Philippines.

Field surveys show that abundance was very low in marine protected sites (MPAs) in mainland Palawan in spite of their proximity to law enforcing bodies. Natural recruits occurred in heavily exploited MPAs, but the fates of released juveniles produced from a decade of artificial propagation are unknown. The breeders have high survival rates in intertidal tanks and were successfully induced to spawn after nearly a year of rearing. The growth rates of hatchery produced juveniles in cages on the reef were as fast as in the wild. Translocated wild trochus had high survival rates but growth rates varied among sites. Elasticity analyses of age-based matrix models revealed that survival of sub- and young adults has the greatest contribution to intrinsic population growth rate, so enhancing the survival of these age groups should be preferred over "headstarting" when conserving trochus.

Efforts to revive the trochus populations should focus on effective long term management/protection of MPAs. Captive rearing of broodstock in subtidal tanks could be a much cheaper alternative to hatchery propagation. Acclimation to predators of hatchery-produced trochus prior to release is hoped to increase their chances of survival. The translocations of wild trochus could be a more effective means of reviving a depleted population in areas having no sign of recruitment.

CONTENTS

LIST OF TABLES

LIST OF FIGURES

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Chapter 1

GENERAL INTRODUCTION

The Philippines along with the Solomon Islands, Malaysia, Papua New Guinea, Indonesia and Timor Leste, which constitute the region called "The Coral Triangle", is regarded as the world's centre of marine biodiversity (Green and Mous 2004; Fidelman et al. in press). This archipelagic region with an area of about 5.7 million km^2 and 153 000 km of coastline (Fidelman et al. in press) holds the world's highest coral diversity of around 500 species (Green and Mous 2004). The Philippines, as part of this region, has more than 17 000 km coastline (BFAR 2008) with 44 000 km^2 of coral reefs (McAllister 1988), of which, 27 000 km^2 of coral reefs were shallower than 10 - 20 fathoms (18.3 – 36.6 m) where reef fisheries occur (BFAR 2008). The Philippines has about 120 endemic fish species of which 31 are marine and a total of more than 3 000 fish species have been reported (Alava et al. 2009), > 3 000 molluscs, > 400 species of scleractenian corals, and nearly 1 000 species of benthic algae (see McAllister 1988; Licuanan and Gomez 2000). The Philippines is therefore regarded as the world's centre of the centre of marine shore fishes with a high degree of endemism (see Alava et al. 2009).

Coral reefs can mitigate climate change and are highly important to the economy of many countries in the tropics and subtropical regions. Coral reefs can protect the coast against coastal erosion (Moberg and Folke 1999), thus enhancing resilience against climate impact. The coral reefs although only cover an approximate 0.1 – 0.5 % of the ocean floor, yet, it harbours at least 1/3 of the world's marine fishes, around 10 % of the fish consumed by humans come from the reef. More than 100 countries have coastline with coral reefs, and more than tens of millions of people depends on coral reefs (see Moberg and Folke 1999; Hodgson and Liebeler 2002). Potential net fishing benefits on the world's coral reefs are estimated at US$ 5.7 billion annually out of total estimated benefits of

US$ 29.8 billion from reefs which include their values for coastal protection, tourism and recreation, and biodiversity value (Cesar et al. 2003). In the Philippines, yield from coral reef fisheries varies depending on coral condition and can range between 3 – 37 metric tonnes of fish per km^2 per year. Reef invertebrates and seaweeds only contribute around 5 % to municipal fisheries, but their contribution to subsistence fisheries could be much higher (see McAllister 1988; White et al. 2000). The Philippine reef fisheries contribute about 11 to 29 % of the country's total fisheries production, and given its archipelagic nature, more than one million Filipinos directly depend on reef fisheries, and about 50 – 60 % of Filipinos' protein intake comes from marine fishes (see White and Cruz-Trinidad 1998; Licuanan and Gomez 2000).

In spite of the huge ecological and economic importance of coral reefs (McAllister 1988; Moberg and Folke 1999; White et al. 2000; Cesar et al. 2003), the Philippine's reef ecosystems have been severely damaged by human action (Gomez et al. 1981; McAllister 1988; Gomez et al. 1994; White et al. 2000; Hodgson and Liebeler 2002; Ablan et al. 2004). In the 1980s, only about 5 % of the country's reef was in excellent condition (Gomez et al. 1981; Gomez et al. 1994; Licuanan and Gomez 2000). Although there is no available record on the recent state of Philippine coral reefs, it is presumed that the trend in percentage of healthy reefs in the country could be in decline due to the unabated illegal fishing practices, outbreak of crown-of-thorns starfish from 2008 – 2010, and coral bleaching in 2010 (Pers. Obs.). Shaish et al. (2010) reported a massive impact of bleaching on their coral nurseries and transplanted coral experiments in 2007 in the Philippines. In the Caribbean, the patterns of reef degradation was attributed by Alvarez-Filip et al. (2009) to mass mortality of grazing urchin and the 1998 coral bleaching event. Reef Check surveys for a total of 34 Philippine reefs between 1997 and 2001 also suggest overfishing (Hodgson and Liebeler 2002). This classes the Philippine coral reefs into the highest category (Stage III) of overfished reef, dominated by immature herbivore fishes and abundance of green filamentous algae on dead corals (Ablan et al. 2004).

As a consequence, efforts to conserve and revive the last remaining reef resources have increased, and a number of marine protected areas (MPAs) have been designated since the 1970s (White and Vogt 2000; White et al. 2005; Weeks et al. 2010). The passage of the Local Government Code in 1991 had caused an exponential increase in the number of MPAs in the Philippines (Weeks et al. 2010). Unfortunately, with the more than 400 MPAs established, only about 20 – 25 % are successful (see Pollnac et al. 2001). In recent years, there are at least 985 MPAs in the country, covering about 14 943 km^2. However, 95 % of these MPAs were community-based which are smaller in size, and would not be sufficient in attaining national conservation targets, unless the numbers and sizes are increased along with the promotion of good governance at each MPA (Weeks et al. 2010). Although a number of successfully managed MPAs have shown recovery of fish species within (White 1986; White and Vogt 2000; Hodgson and Liebeler 2002; Russ et al. 2004; White et al. 2005) and outside their boundaries (Russ et al. 2004; Abesamis and Russ 2005; Stockwell et al. 2009), there was limited recovery of invertebrates in MPAs (Hodgson and Liebeler 2002; Raymundo 2003; Maliao et al. 2004; Gomez and Mingoa-Licuanan 2006; Lebata-Ramos et al. 2010).

Marine Protected Areas (MPAs) which are of various types are a form of resource management that aimed to protect part or the entire enclosed environment from human action (Sale et al 2005; Sumaila et al. 2000). In England, MPAs are of five designations depending on purpose: 1. “Special Areas of Conservation (SASs) – designated for marine habitats or species of European importance”, 2. “Special Protection Areas (SPAs) – designated to protect populations of specific species of birds of European importance”, 3. “Sites of Special Scientific Interests (SSSIs) – This include the most spectacular and beautiful habitats teemed with wildlife”, 4. “RAMSAR Sites – are sites for wet land birds that are protected internationally” and 5. “Marine Conservation Zones (MZCs) – designated to protect nationally important marine wildlife and their habitats” (NatureEngland 2011). In the Philippines, large MPAs like the Tubbataha Reefs Natural Park which covers an area of about 968 km^2 is of international significance having been designated both a RAMSAR and World Heritage Sites (TMO 2011). When properly managed, strictly no-take MPAs can help revive the population of the previously overfished fishery resource and help sustain the neighbouring unprotected fishing grounds through spill-over effects (Sumaila et al. 2000; Abesamis et al. 2005). In the Philippines, many MPAs are small and are managed at the community level (Weeks et al. 2010). Factors that influenced the success of these community-based MPAs in the include: "population size of the community, a perceived crisis in terms of reduced fish population, successful alternative income projects, high level of participation in community decision making, continuing advice from the implementing organisation, and inputs from local government" (Pollnac et al. 2001). However, most community-based MPAs in the Philippines are inadequately managed (Weeks et al 2010). Partly protected MPAs have not shown the recovery of any fishery resource, much more for MPAs where fishing is unhampered. In New Zealand, long term protection (for nearly 30 years) of marine parks have resulted to an increase in legal-sized lobster of 11 times more abundant and biomass 25 times higher in the no-take marine park following park establishment, while there was no significant change in lobster abundance in partially protected marine parks (Shears et al. 2006). The long term protection of the Tubbataha Reefs Natural Park in the Philippines has also made trochus to recover reaching an average density of about 6 000 ind ha^{-1} in areas where they abundantly occur, yet poaching for two years had reduced their abundance by 70 % (Dolorosa et al. 2010; Jontilla et al. 2011; Jontilla et al. in press).

To conserve and effectively manage a coastal fishery resource, releasing of hatchery produced juveniles have been practiced with the following objectives as outlined by Bell et al. (2008b): 1) "to restore severely depleted spawning biomass to a level where it can once again provide regular, substantial yields" otherwise known as **Restocking**; 2. "to augment the natural supply of juveniles and optimize harvests by overcoming recruitment limitation" (**Stock enhancement**) and 3) "the release of cultured juveniles into unenclosed marine and estuarine environments for harvest at a larger size in put, grow, and take operations" (**Sea ranching**). In

Japan, the success in releasing hatchery cultured juveniles of 34 species of finfish, 12 species of crustaceans, 25 species of shellfish and 8 other species to augment fishery production varied according to species. The success in stock enhancement of salmon in Japan was attributed to environmental factors, improved ranching techniques, and high recaptures rates. Another species of salmon (Masu salmon) however have lower recapture rates suggesting that it is more practical to protect the breeding adults and their spawning ground over the release of hatchery produced juveniles. Success in scallop sea ranching have been attributed to "sufficient supply of high-quality natural seedlings stemming from development of natural seed-collection techniques"; removal of the starfish predators in the area of release; "management of stock by crop rotation", and active participation of the cooperatives in the entire process of stock enhancement starting from the procurement of seedlings up to harvesting, processing and marketing. The success in restocking of flounder have been attributed to food abundance, size of juveniles at release, timing of release to coincide with summer months which promote faster growth of the seeded stocks thus reducing high impact of predation, intense fishing effort and the tendency of the released stock not to disperse largely (see Masuda and Tsukamoto 1998).

The general consequences of biodiversity loss and overfishing can lead to the coral reefs' physical breakdown (McClanahan 1995; McClanahan et al. 1996) with the precise consequences varies depending on the level of importance and the numbers of the species, and composition of the communities from which they are lost (Crowe 2005). Stachowicz et al. (2007) provided a comprehensive review on the effects of marine biodiversity on communities and ecosystems. Overfishing along with other factors can cause macroalgal outbreak (Williams et al. 2001; Hodgson and Liebeler 2002; Ablan et al. 2004; Aronson and Precht 2006) which can lead to increased coral mortalities (Birrell et al. 2005; Aronson and Precht 2006) and prevent the settlement of other rock dwelling animals like barnacles (Jernakoff 1985). The way therefore to prevent the outbreak of filamentous algae and coral preying organisms is through the revival of a diverse reef fauna especially those organisms that can help control the outbreak of coral competitors and predators. Grazing gastropods can control epiphytic and macro algal bloom (Pace et al. 1979; Geller 1991; Jernakoff and Nielsen 1997; Jenkins and Hartnoll 2001; Hily et al. 2004). Under laboratory condition, Dwiono et al. (1997) reported that 20 000 hatchery produced *Trochus niloticus* juveniles (4 – 7 mm in diameter) can consume sessile diatoms covering an area of 6.5 m^{-2} within a week.

Like reef fishes, the reef invertebrate fishery forms a substantial contribution to the economies of coastal communities especially in the Indo-Pacific (Nash 1993; ICECON 1997; Hahn 2000; FAO 2010), yet, these species have been overexploited, requiring the need to effectively manage the remaining populations (Nash 1993; Hodgson and Liebeler 2002; Bell et al. 2005; Bell et al. 2008a; Bell et al. 2008b; FAO 2010). Species of sea cucumbers and macrobenthic molluscs are the two most important groups of invertebrates that have been extensively exploited on the reefs. For example, the sea cucumber fishery in the Indo-Pacific is multi-species with 20 – 30 species being exported per country, or at least 60 species are harvested in more than 40 countries. The annual world's harvest of sea cucumber is about 100 000 tonnes of live animals. In 2000, export volume of processed sea cucumbers to Asian Market was about 6 000 tonnes, valued over US$ 130 million (FAO 2010; Purcell 2010). Harvesting of sea cucumber is managed by licensing and use of allocated plots in some developed countries like Canada and Australia. In a developing country like the Philippines, however, the sea cucumber fishery is open to all fishers, where they collect even the small individuals as they believe that other fishers will also collect them if left behind (FAO 2010). As a consequence, populations of high valued sea cucumbers species in the Pacific have been overfished, and fishers have shifted to the collection of low valued species (Lovatelli et al. 2004; FAO 2010). The Philippines was one of the leading exporters of sea cucumbers ten years ago, yet this trend has changed in recent years as a result of overfishing and the development and expansion of sea cucumber fisheries in other countries (FAO 2010). Efforts to maintain the supply for the increasing demand for the species include marine aquaculture and sea ranching but is limited to a few species and localities in the Philippines (Bell et al. 2008a; FAO 2010; Purcell 2010).

Other overly exploited reef invertebrate species include the giant clams (*Tridacna* spp. and *Hippopus* spp.), considered as the most endangered of all reef invertebrate species. Because of their large size and shallow habitats, overfishing had resulted to a collapse or local extinction of their natural populations (Basker 1991; Gomez and Mingoa-Licuanan 2006; Richter et al. 2008; bin Othman et al. 2010). All seven species of giant clams are listed under IUCN Red List of Threatened species (IUCN 2010), under Appendix II of CITES (CITES 2011), and listed as endangered species in the Philippines (DA 2001; BFAR 2011), yet recent efforts to conserve their populations is restricted to the restocking of mass produced single clam species (*Tridacna gigas*) in a few localities (Cabaitan et al. 2008; Lebata-Ramos et al. 2010). A long term (almost 20 years) mass production and reseeding of six giant clams species (but generally focused on single species) have been conducted by Gomez and Mingoa-Licuanan (2006) from 1985 – 2005, releasing about 80 000 juveniles in > 40 sites in > 30 % of the 62 coastal provinces of the Philippines. Although some of the released clams have reached maturity, the success of the project was particularly challenged with problems related to community cooperation and involvement.

Abalone (*Haliotis* spp.) are also overexploited but have been conserved by mass production, cage culture and sea ranching in some parts of the Philippines (Capinpin et al. 1998; Capinpin et al. 1999; Maliao et al. 2004; Lebata-Ramos et al. 2010). The spiny lobster and triton shell have also been overexploited (Hodgson and Liebeler 2002), but unlike the previously mentioned reef invertebrates species, there is no known mass production and sea ranching program for these species in the Philippines. Spiny lobsters (*Panulirus* spp.) are heavily exploited for the live fish trade while the shells of tritons (*Charonia tritonis*) are collected because they make a good souvenir item. The triton shells (*Charonia* spp.) are used as one of the global reef health indicators because of their beautiful shells that are easy to collect. The triton is a protected species on the Great Barrier Reef because of their low numbers and their role in preying on coral predators: crown-of-thorns sea stars (Hodgson and Liebeler 2002). In the 1960s, about 10 000 tritons per year were harvested from the Great Barrier Reefs, which has potentially reduced their population and their predation on the crown-of-thorns sea stars (see Hodgson and Liebeler 2002 and references therein).

Just like other fishery resources, overfishing has resulted in a population decline of the commercially important large reef gastropod *Trochus niloticus* or trochus (Nash 1985; Nash 1988; Nash 1993; Nash et al. 1995; Smith et al. 2002; Purcell et al. 2004; Bell et al. 2005; Lasi 2010). Trochus are known to occur in high densities on intertidal reefs (Nash 1985; Nash1993; Tsutsui and Sigrah 1994; Nash et al. 1995; Dolorosa et al. 2010), so overexploitation leading to the removal of a large portion of its population could have a negative impact both on fishers and the coral ecosystems. Trochus are an important source of protein and revenue for coastal dwellers (Nash 1985; Nash 1988; Nash 1993; Nash et al. 1995; Amos 1997; Crowe et al. 1997; Hahn 2000). They have been regarded as the most economically important gastropod species in the Indo-West Pacific Region (Heslinga and Hillmann 1981; Carpenter and Niem 1998) owing to the high demand and value for their shells (Nash 1993; ICECON 1997). Trochus have a patchy distribution (see Nash 1993; Carpenter and Niem 1998; Bell et al. 2005) because of their short larval period resulting in limited larval dispersal (Heslinga and Hillmann 1981), but successful introduction within and outside the natural range of the species (Gillett 1993; Ponia et al. 1997; Bell et al. 2005) to cater for the increasing demand for its shell, have expanded its distribution. Trochus is long-lived (> 20 years old), can reach a large size of 160 mm, and has good shell quality, making it the only species in the family Trochidae that is extensively exploited for pearl buttons and a large variety of products (Heslinga and Hillmann 1981; Burhanuddin 1997; Carpenter and Niem 1998; Ramakrishna et al. 2010). The estimated annual harvest of trochus in the 1980s was between 5 000 and 6 000 tonnes with a dockside value of about US$ 4 million, but the retail value of buttons alone was exceedingly high, reaching more than US$ 200 million (see Heslinga et al. 1984; Hahn 2000; Bell et al. 2005).

The overexploitation of trochus (Burhanuddin 1997; Carpenter and Niem 1998; Hodgson 1999; Smith et al. 2002) has forced fishers to venture in deeper waters (Rao 1937; Etaix-Bonnin and Fao 1997; Ledua et al. 1997), or to more offshore reefs (Nash 1985), encroaching even e protected areas or crossing international boundaries (Rao 1937; Stutterd and Williams 2003; Dolorosa et al. 2010; Ramakrishna et al. 2010; Jontilla et al. in press). Prior to its commercial exploitation, trochus were plentiful in intertidal areas (Talavera and Faustino 1931; Rao 1937; Etaix-Bonnin and Fao 1997). Reported densities for unexploited populations were high, ranging between 2 to 33 ind m^{-2} (or 20 000 – 330 000 ind ha^{-1}) in Yap and 1 – 25 ind m^{-2} (10 000 – 250 000 ind ha^{-1}) or even as high as 66 ind m^{-2} (660 000 ind ha^{-1}) in the Cook Islands (see Tsutsui and Sigrah 1994). In Tubbataha Reefs Natural Park, Philippines, the highest trochus density record was at 11 000 ind ha^{-1} (Dolorosa et al. 2010).

Trochus has been used as a reef overfishing indicator for the coral reef fishery (Hodgson 1999). Trochus overfishing was reported shortly after the start of its commercial harvest in the early 1900s (Bell et al. 2005). Overfishing was reported in countries such as Japan (Isa et al. 1997), Vietnam (Hoang et al. 2007), India (Ramakrishna et al. 2010), Indonesia (Burhanuddin 1997), Philippines (Heslinga and Hillmann 1981; Gapasin et al. 2002), Fiji (Ledua et al. 1997) New Caledonia (Etaix-Bonnin and Fao 1997), and the rest of those small island countries in the south Pacific (Cheneson 1997; Leqata 1997) ; Micronesia (Fanafal 1997), and Australia (Magro 1997; Ostle 1997; Stutterd and Williams 2003). In overexploited areas, the density of trochus could be as low as 2.2 – 3.3 ind ha^{-1} as on the case of Cartier Reef in Australia (Smith et al. 2002), or as low as 0.42 ind ha^{-1} in Indonesia's National Marine Park in southeast Sulawesi (Burhanuddin 1997). In worst cases, isolated extinction could have occurred in some reef areas.

When overexploited, trochus may fail to recover, or recovery can be extremely slow compared to fish, because of their limited mobility, short larval period and tendency to settle near to the parent population (Nash 1993). As a consequence, the trochus fisheries have been managed in many different ways including the use of size limits, closed seasons, marine sanctuaries, quotas, restocking with hatchery produced juvenile and translocation (Heslinga and Hillmann 1981; Heslinga et al. 1984; Gillett 1989; Gillett 1993; Nash 1993; Amos 1997; Crowe et al. 1997; Foale and Day 1997; Lee and Toelihere 1997; Amos and Purcell 2003; Stutterd and Williams 2003; Purcell 2004; Ramakrishna et al. 2010). These have had varied levels of success as the intensity of fishing, nature of remaining stock, environmental conditions, and level of management varies between countries or even within reefs (Gillett 1989; Gillett 1993; Nash 1993; Amos 1997; White and Vogt 2000; Raymundo 2003; Bell et al. 2005; Dolorosa et al. 2010; Purcell and Cheng 2010; Ramakrishna et al. 2010). Trochus management has been reported as more successful at a smaller scale such as community-managed reserves rather than across a large geographic area (ICECON 1997; Dumas et al. 2010) suggesting that a different management approach for each marine reserve may be required depending on the status of the natural stock, availability of habitats and community involvement.

Trochus is not a CITES listed species but because of overharvesting, its exploitation had been prohibited in countries like Indonesia (Dwiono et al. 1997), Vietnam (Hoang et al. 2007) and India (Ramakrishna et al. 2010). It had also been listed as an IUCN commercially threatened reef invertebrate (Heslinga et al. 1984; Hahn 2000). In the Philippines, the species is protected under the Fisheries Administrative Order 208, series of 2001 (DA 2001; Floren 2003; BFAR 2011). But prior to that, the following number of laws have been passed (BFAR 2011) to manage the trochus fisheries along with other harvested species:

1. In the early 1900s it was prohibited to harvest trochus under 8 cm in diameter (Hedley 1917).

2. Fisheries Administrative Order No. 19, Series of 1939 – provides guidelines on price of shells (*T. niloticus* was at PhP 0.25 kg^{-1}) and pertinent policies such as inspection of marine mollusc shells prior to exportation.

3. Fisheries Administrative Order No. 69, Series of 1963 – Provided a set of regulations governing the gathering of ornamental or fancy shells, sea snakes, trepang (dried sea cucumbers), corals and miscellaneous minor aquatic products. Entities with commercial permit to gather or trade the species are required to make quarterly report of the quality and quantity of collected or traded species.

4. Section 11 of the Republic Act No. 8550, otherwise known as the Philippine Fisheries Code of 1998, calls for the protection of rare, threatened and endangered Species. Section 97 prohibits the collection, harvesting or gathering of rare, threatened or endangered species as listed in the CITES and as determined by the Department.

5. Fisheries Administrative Order No. 208, Series of 2001 – listed *Trochus niloticus* as threatened species.

6. Republic Act No. 9147 – "An act providing for the conservation and protection of wildlife resources and their habitats, appropriating funds thereof and for other purposes" was enacted in 2001.

In this study, we assessed the condition of trochus population and the on-going conservation measures in Palawan, Philippines to find some cost-effective ways in reviving the trochus populations in the country's depleted reefs.

Thesis outline

This thesis on the status and conservation of *Trochus niloticus* in the Philippines is divided into six parts, focused on answering the following questions: (1) How does exploitation affect the population of trochus in marine protected areas? (2) How effective was the trochus breeding and reseeding program in Palawan? (3) How do trochus broodstock kept in intertidal tanks respond to induced spawning? (4) How can growth and survivorship of trochus during intermediate culture be improved? (5) Will trochus grow and survive well when translocated? and (6) At which age stages of trochus should conservation and management focus on?

In **Chapter 1**, we provided an overview of the research. In **Chapter 2**, we described how continued exploitation in marine protected areas (MPAs) has affected the population of trochus. We also compare the spatial distribution of trochus in different habitat types in exploited and unexploited MPAs. **Chapter 3** examines the success and failure of trochus breeding and reseeding initiatives conducted both by private and government organisations in Palawan. In **Chapter 4**, we examine the possibility of rearing and induced breeding of wild adult trochus held captive for almost one year in intertidal tanks. We examine the realised fecundity, fertilisation rates and hatching rates and relate it with the reproductive output of newly collected breeders from the wild that were induced to spawn. In **Chapter 5**, we explore the growth and survival of trochus juveniles for both indoor and subtidal intermediate culture. In particular, we explore the use of coconut leaves as substrate and the potential of rearing trochus in deep reef cages. **Chapter 6** investigated the growth and survival of wild trochus juveniles and adults when translocated in heavily exploited and unexploited areas where few numbers of trochus were found. Specifically, we monitored the growth and survival of translocated trochus in comparison with those found in their natural habitat. **Chapter 7**, models the elasticity of populations of trochus to changes in the survival and fecundity of different age stages of trochus. In this chapter, we used a combination of field and secondary data to perform elasticity analyses in order to assess the age classes on which conservation should focus on. Finally, the **Chapter 8** or **Concluding Remarks** presents the key findings in this research, limitations and its implications for trochus management and future research in the Philippines.

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Chapter 2

SPATIAL AND TEMPORAL ABUNDANCE OF *Trochus niloticus* IN MARINE PROTECTED AREAS IN PALAWAN, PHILIPPINES: PROSPECTS FOR CONSERVATION

Roger G. Dolorosa, Alastair Grant and Jennifer A. Gill

ABSTRACT

Trochus niloticus, a Philippine threatened reef gastropod, was once an important fishery resource for millions of marginal fishermen and a significant source of revenue. Commercial harvest began in the early 1900s and by early 2000, unregulated harvesting has decimated most trochus population, prompting the government to declare it a threatened species. Illegal exploitation however, continued to threaten its population even at Tubbataha Reefs Natural Park (TRNP). Conservation activity has mainly focused on the costly and ineffective release of hatchery produced juveniles in areas where gleaning is uncontrolled. The abundance of trochus was surveyed at a number of Marine Protected Areas (MPAs) in Palawan, Philippines to determine its status, and to propose more relevant conservation measures. In unguarded and continuously exploited MPAs, trochus were present at extremely low numbers and were smaller in sizes than in well guarded or unexploited sites. Only empty trochus shells were encountered in the TRNPs most distant part (from the Ranger Station), the Jessie Beazley Reef, suggesting either recruitment failure or localised extinction due to overharvesting. Recruitment was occurring at exploited areas at Palawan on the mainland suggesting that effective long term protection of these sites can help revive trochus populations. For a well managed MPAs showing no sign of trochus recovery, the translocation or aggregation of wild adult trochus as an initial breeding population is expected to hasten recovery. The restoration of reef invertebrates in a network of MPAs can help maintain a healthy reef ecosystem, which can result in better fishing, ecotourism, shoreline protection and a wide array of ecological and economic benefits.

Keywords: Abundance, size structure, protected area, harvesting, *Trochus niloticus*

INTRODUCTION

Marine protected areas (MPAs) as a conservation strategy is becoming an important tool in conserving the depleted marine resources as it can "potentially generates a wide range of consumptive use, non-consumptive use and non-use values that include: critical habitat protection, conservation of marine biodiversity, recovery of threatened and endangered marine species, increased recreational benefits and increased biomass of harvested marine species" (Grafton et al. 2011). In the Philippines, few numbers (< 1 000 km^2 in area) of marine protected areas (MPAs) designated in the 1970s (White and Vogt 2000; White et al. 2005; Weeks et al. 2010), have increased exponentially after the passage of the Local Government Code in 1991, reaching 985 (about 15 000 km^2 in area) in 2008 (Weeks et al. 2010). Unfortunately, many of these MPAs are poorly managed and only about 20 – 25 % are successful (see Pollnac et al. 2001).

The importance of understanding the status of a resource prior to proposing any management scheme cannot be underestimated. Annual recording of the abundance trend for example can provide evidence on how known or unknown variables or phenomena are affecting the studied population. One widely known resource assessment activity is the Reef Check, designed to assess the global, regional or local conditions of the reef (Hodgson 1999; Hodgson 2002; Hodgson et al. 2004). This assessment method provides researchers and conservation workers with information on the coral reefs' health and coral reef fisheries, useful in predicting trends with or without management interventions. Aside from trends of abundance, stock assessment can provide a wide array of biological data such as age structure, species composition, fecundity, sex ratio, mortalities, preferred habitats, and growth, to mention a few (Nash 1985; Nash et al. 1995; Smith et al. 2002; Trianni 2002; Guerra-García et al. 2004; Araújo et al. 2005; Shears et al. 2006; Pakoa et al. 2010).

Trochus niloticus, commonly called trochus, is a commercially important and overly exploited large reef gastropod with patchy distribution in the Indo-Pacific Region (Carpenter and Niem 1998; Lemouellic and Chauvet 2008). The short larval period of trochus (Heslinga and Hillmann 1981) and the isolation of many reefs were thought to be the main reasons for its inability to colonise suitable offshore reef habitats (Nash 1993; Bell et al. 2005), unless translocated. In more than 6 decades, trochus translocation to about 60 places within the Indo-Pacific Region has expanded its distribution, eventually becoming an important fishery resource at many of the locations to which it has been introduced (Sims 1984; Gillett 1993; Zoutendyk 1997; Trianni 2002; Pakoa et al. 2010).

Harvesting of trochus for a variety of uses started in prehistoric times (Landberg 1966, Landberg 1968; O'Connor et al. 2002; O'Connor and Veth 2005; Anonymous 2007), but populations only began to decline when commercial harvest of its shells for the mother-of-pearl button industry started in the early 1900s (Rao 1937; Heslinga and Hillmann 1981; Nash 1985; Nash1993; Bell et al. 2005). The sharp decline in the volume of export (Hahn 2000), and abundance (Rao 1937; Nash 1993; Bell et al. 2005) of trochus shortly after the start of its commercial harvest, suggest its high vulnerability to exploitation because of its large size freely exposed on the reef, relative immobility and well defined shallow reef habitat (Nash 1985; Nash 1993; Nash et al. 1995).

In their natural habitats, trochus are associated with seaward reef areas subjected to strong wave action (Nash 1985; Nash 1993), although in some areas, they were plentiful in leeward reefs with wide suitable habitat (Smith 1987). Because of their tendency to aggregate (Moorhouse 1932; Rao 1937), trochus density has been reported to reach as high as 66 ind m^{-2} or 66 000 ind ha^{-1} (Tsutsui and Sigrah 1994) or as low as 0.42 ind ha^{-1} when overexploited (Burhanuddin 1997). The overexploitation of trochus in many countries had led to efforts to revive its population through varied conservation measures (Nash 1993; Lee and Lynch 1997; Bell and Gervis 1999; Bell et al. 2006).

Monitoring the status of trochus or other harvested marine life is an important step in assessing the effectiveness of management measures, and in providing a basis for sound conservation decision making (Sims 1984; Foale and Day 1997; Zoutendyk 1997; Foale 1998; Jennings 2001). In India, for example, a recent extensive survey on trochus population following a 10-year ban on harvest suggested that it was feasible to open the fisheries for three years (Ramakrishna et al. 2010). In the Cook Islands, periodic population monitoring of density and standing stock of the introduced trochus was conducted to assess population recovery from the previous harvest and to estimate catch quota for the next harvest (Sims 1984; Nash et al. 1995; Zoutendyk 1997). The assessment of trochus in Tongatapu Lagoon, Kingdon of Tonga, 12 years after its introduction, suggests successful recruitment and establishment of a population but harvesting was not recommended for a further 8 years to allow increase in population size (Pakoa et al. 2010). Size measurement along with abundance surveys are essential to assess recruitment overfishing, which can be detected if populations are dominated by sizes smaller than the allowed harvestable sizes, or sizes that are less likely to be searched or exploited (Nash 1993). Population assessment and the implementation of varied conservation measures have taken place in most trochus producing countries only after harvests have begun to fall (Nash 1985, Nash 1993; Burhanuddin 1997; Dwiono et al. 1997; Magro 1997; Ostle 1997; Bell et al. 2006; Bell et al. 2008).

In the Philippines, except for the trochus monitoring in Tubbataha Reefs Natural Park (TRNP) from 2006 (Dolorosa et al. 2010; Jontilla et al. in press) up to the present, no comprehensive study about trochus exploitation has been conducted in spite of having been recognised as one of the world's top trochus producers (Hahn 2000) with the highest quality (Carleton 1984; Nash 1985). Often, trochus is reported as a small part of fishermen's harvest (Schoppe et al. 1998; Del Norte-Campos et al. 2003), suggesting an extremely low abundance. In fact, as early as 1980, trochus had already been overfished in the Philippines (Heslinga and Hillmann 1981). The declaration of trochus as a nationally threatened species in 2001 (DA 2001; Floren 2003), spawned interests on stock enhancement through the release of artificially produced juveniles as a conservation measure (Gapasin et al. 2002; Gonzales et al. 2009; Avillanosa et al. 2010), without firstly looking into its status in the wild. The mass production of trochus juveniles for release is costly and ineffective compared to the traditional methods of species management (Amos 1997), and should not be used when there are sufficient wild adults for translocation (Bell et al. 2005).

A number of conventional methods could be used to conserve the trochus populations (Nash 1993; Nash et al. 1995; Bell et al. 2005). Deciding on which one to use requires prior knowledge about the current status of trochus population. In this study, we assessed the status of trochus in exploited and unexploited MPAs in Palawan, Philippines. Specifically, we determined how the abundance of trochus varies across types of habitat and sites and compared the size structure of trochus in different habitats across the studied sites.

METHODS

Study sites

The study covered six sites in three localities in Palawan, Philippines from which 376 transect lines (2 x 20 m) were surveyed. Four sites were in the Tubbataha Reefs Natural Park (TRNP) in the middle of the Sulu Sea (Sites 1 – 4); and one each at Rasa Island in the southern part of Palawan (Site 5), and Binduyan (Site 6), a northern village in the city of Puerto Princesa (**Figure 2.1**).

TRNP lies within 8°43'– 8°57' N and 119°48'–120°3' E, about 150 km southeast of Puerto Princesa City, Palawan and 130 km south of the municipality of Cagayancillo. In total, the park covers an area of 96 828 ha (see Dolorosa et al. 2010) of which roughly 10 000 ha are coral reefs (Arquiza and White 1999). Detailed descriptions of the North and South Atolls are provided in White and

Vogt (2000) and Palaganas et al. (1985). Several incidence of poaching have been reported from TRNP and trochus abundance have been greatly reduced from 2006 until 2009 (Dolorosa et al. 2010; Jontilla et al. 2011; Jontilla et al. in press) except in areas that are relatively close to the Ranger Station. In such a case, TRNP was subdivided into four study sites: (Site 1) Unexploited North Atoll, composed of three stations near the Ranger Station; (Site 2) Moderately Exploited North Atoll, composed of five stations that are about 9 km away from the Ranger Station; (Site 3) Heavily Exploited South Atoll, composed of five stations around the South Atoll Reef (about 14 km from the RS); and (Site 4) the distantly located Jessie Beazley Reef situated 24 km away from the RS (**Figure 2.1, Table 2.1**). A station is referred in this study as a specific place where at least two or three types of surveyed habitats occurred adjacent to each other.

Table 2.1. The four study sites in TRNP and their proximate distance from the Ranger Station, and the two study sites in the mainland Palawan: Rasa Island – about 1 km from the mainland Palawan and the two reefs in Binduyan which are connected to the mainland Palawan. *One station from Site 1 was excluded because of absence of trochus possibly due to sand shifting. ** Remarks were based on the results of previous surveys of Dolorosa et al. 2010; Jontilla et al. 2011 and Jontilla et al. in press).

Site Number and Name	Mean (±sd) distance (km) of stations from the Ranger Station		Remarks**
1 (North Atoll, TRNP)	2.25 ± 2.63	(n=2)*	Unexploited
2 (North Atoll, TRNP)	8.95 ± 3.44	(n=5)	Moderately exploited
3 (South Atoll, TRNP)	14.18 ± 1.87	(n=5)	Overexploited
4 (Jessie Beazley Reef, TRNP)	24.01	(n=1)	Overexploited
5 (Rasa Island, Palawan)			Overexploited
6 (Binduyan, Puerto Princesa City)			Overexploited

Rasa Island (9°13'21.25" N and 118°26'38.06" E), referred here as Site 5, has an area of 834 ha, situated just 1 km offshore of the Municipality of Narra, Palawan (**Figure 2.1**) and is therefore accessible even to fishers with small paddled canoe. The island has 175 ha of coastal forest, 560 ha mangrove, 39 ha coconut plantation and 60 ha barren or sparsely vegetated sand and coral outcrops (Widmann et al. 2010). The topographic map from the GIS section of the Palawan Council for Sustainable Development (PCSD), however, suggests a much wider shallow coast (156 ha) and coralline rocks (198 ha) and coral reefs (27 ha). The island, as home to a restored dense population of the critically endangered Philippine Cockatoo, is a popular bird watchers' destination. In 2006, the island was declared a Wildlife Sanctuary through Presidential Proclamation 1000 (Widmann et al. 2010), but fishing and gleaning remained unregulated. Fishing activities include the use of gill net, fish corral and spear gun. Aside from fishing, gleaning is a common activity around the island. During one of our visits at the island, around 25 gleaners were noted searching and collecting seaweeds and invertebrates at the exposed intertidal area. Informal conversation with some fishermen suggests that the island is fished on a daily basis and therefore this site is referred here as overexploited MPA.

Binduyan (10^o 00'46.78" N and 119^o 04'14.92" E), referred to as Site 6, is a coastal and the second northernmost village of Puerto Princesa City (**Figure 2.1**). Two small adjacent reefs near the village were surveyed. These were the Sabang Reef marine sanctuary – a seeding site for hatchery produced trochus juveniles (**see Chapter 3**) and Bitauran Reef - an open-accessed area. Sabang Reef is part of a 40-ha village reef sanctuary established in 2003 through City Ordinance 192. Sabang Reefs' exposed area (~14 ha) at low tide is nearly twice larger than that of Bitauran Reef (~8.5 ha). The reefs are about 500 m apart, both extending roughly 300 m from the shore, forming a small shallow cove in between. Being connected to the mainland Palawan, both reefs are highly accessible to gleaners of all ages, collecting any edible marine life they found on the reef during low tide. During one of our visits, there were about 20 gleaners at each reef. These two reefs in Binduyan are also referred to as heavily exploited reefs. Both reefs exposed at low tide were characterised by flat coral rocks at the inner and central reef flat, and with large loose coral boulder along the outer margin.

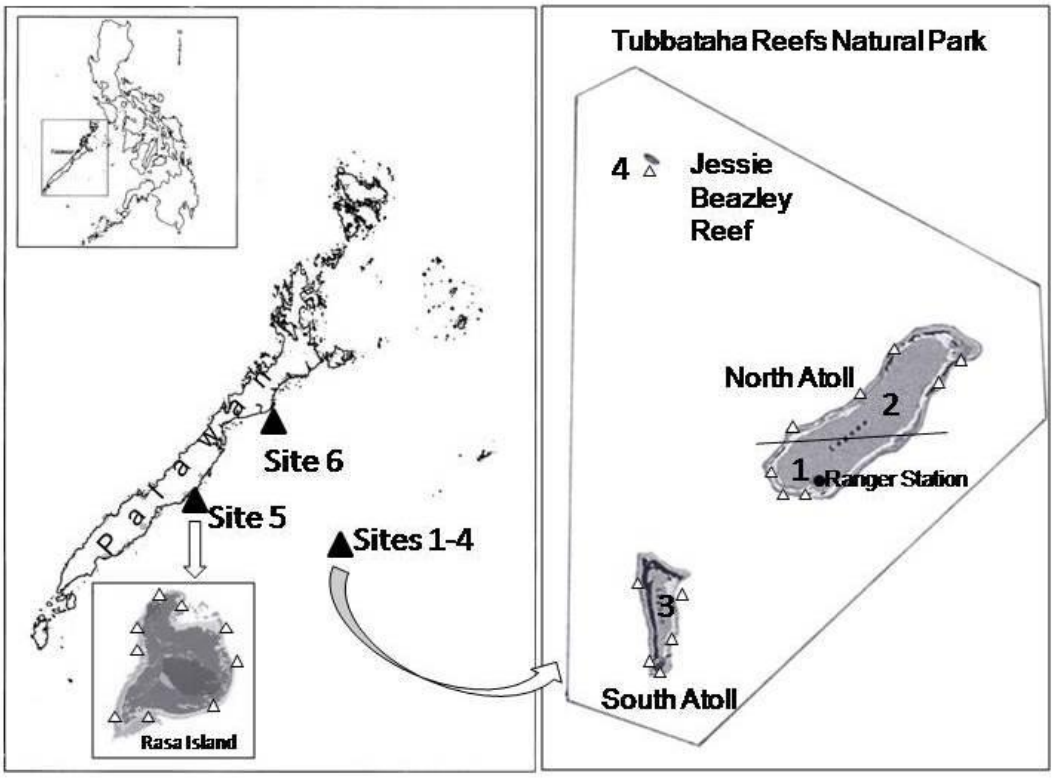

Figure 2.1. The six study sites.1 – Unexploited North Atoll; 2 – Moderately Exploited North Atoll ; 3- Overexploited South Atoll; 4 – Overexploited Jessie Beazley Reef; 5 – Overexploited Rasa Island; 6 – Overexploited Binduyan and part of Green Island Bay. Sampling stations are indicated by small triangles (i.e. around Rasa Island and in Atolls at TRNP).

Three types of habitats such as flat, boulder and complex were surveyed at each site except in Site 4 where only the boulder and complex habitats were surveyed (**Table 2.2**). The characteristics of the three types of habitats were as follows: **Flat Habitat** at TRNP was the reef crest or the highest part of the reef. This is the first part of the reef to be exposed at low tide and the last to be covered with water at high tide, thus providing an extremely variable condition for any settling organism. The substrate was generally composed of rubble or loose fragments of branching corals (few cm up to about 6 cm), and some scattered flat coral rocks, which do not retain water at low tide. This narrow strip of rubble bordered the reef between the lagoon and the seaward reef slope in an atoll as for the case of North and South Atolls of TRNP (**Figure 2.2, left)**. In the mainland Palawan, surveyed flat habitats were found surrounding the island, usually found between the mangrove forest or seagrass beds and the boulder habitats. Flat habitats surveyed at the mainland Palawan differs from TRNP for having a flat rocky surface with turf seaweeds, and can hold a small amount of water even during low tide.

Boulder Habitats were composed of loose fragments of rocks that are mostly larger than 20 cm in diameter. Boulder habitat is generally composed of a narrow strip of dead massive and sub-massive corals found along the edge of the reef flat, and is the last to be exposed during the lowest low tide, or first to be submerged when the tide starts to rise (**Figure 2.2, centre**).

Complex Habitats were found at the seaward reef slope, submersed all the time to a depth of 3 – 5 m deep during the highest high tide. This habitat was characterised by a mixture of dead and live corals of various life forms, and serves as refuge for trochus and other coral dwelling organisms (**Figure 2.2, right**).

Figure 2.2. The three types of surveyed *T. niloticus* habitat in MPAs in Palawan, Philippines. Flat habitat at TRNP (left), and boulder (middle) and complex (right) habitats at Rasa Island.

Abundance and size structure of *Trochus niloticus* in unexploited and exploited sites

The abundance and size structure of trochus were determined by using the Reef Check method in assessing reef invertebrates (Hodgson et al. 2004). This was modified by reducing the width of the transect lines into 2 m instead on the standard 5 m. This was done to increase the number of stations and to exhaustively search and measure the trochus found along each transect at a shortest time possible because the survey was timed when the reef was exposed or when the water was calm. A reconnaissance survey for the presence of trochus was firstly conducted before laying the transect lines. This was done by walking on the exposed reef or snorkelling at the subtidal habitats, searching for trochus and other reef associated invertebrates under rocks. Depending on the area and habitat availability at each station, four or eight 2 x 20-m transect lines were laid at least at 5 m intervals parallel to the coast line. The number of transects laid per habitat was patterned with Reef Check where a 100-m transect line is composed of four 20-m long segments laid at 5-m intervals on the edge of the reef parallel to the shoreline (Hodgson et al. 2004). In this instance however, the term segment in Reef Check is referred in this study as transect. Substrate types (e.g. hard coral, rock, rubble, sand) were recorded at 0.5 m intervals of transect lines, following the substrate criteria and point intercept method described by Hodgson et al. (2004).

To determine the habitat complexity or Rugosity Index of the reef, a weighted 20-m rope (10 mm diameter) which follows the contour of the reef was placed parallel the 20-m transect line (**Figure 2.2**). The Rugosity Index is expressed as the ratio between the total length of the 20-m weighted rope and the straight distance covered by the weighted rope that follows the contour of the reef. Thus, the Rugosity Index of flat habitat is nearly one or lower compared to the Rugosity Index of boulder and complex habitats (see Alvarez-Filip et al. 2009). Data collection was conducted from November 2009 until July 2010.

In total, 376 transect lines (2 x 20 m) were surveyed in 25 stations within six sites (**Table 2.2**) covering an area of 15 040 m^2. The transect lines were distributed among the three types of habitats at 26 % (flat), 41 % (boulder), and 33 % (complex) (**Figure 2.3).** Except in Site 3, all flat habitats were surveyed by reef walking. Reef walking was also used to assess the boulder habitats in Site 1, and all boulder habitats in Sites 5 and 6. All complex habitats were surveyed by snorkelling during low tide. All trochus found along each transect were counted and measured for their maximum basal diameter to the nearest mm with the aid of a ruler glued on a slate board.

Table 2.2. Numbers of 2 x 20 m transect lines per study site and types of habitat surveyed. A – flat, B – boulder, C – complex, UE – unexploited, ME – moderately exploited, OE- overexploited.

Site Number	Site Name	Status	Number of Stations	Habitat Type A	B	C	Total Transects
1	North Atoll, TRNP	UE	3	20	16	20	56
2	North Atoll, TRNP	ME	5	16	36	36	88
3	South Atoll, TRNP	OE	5	8	32	16	56
4	Jessie Beazley Reef, TRNP	OE	1	0	4	4	8
5	Rasa Island, Narra, Palawan	OE	9	44	48	36	128
6	Binduyan, Puerto Princesa City	OE	2	8	20	12	40
	Total		25	96	156	124	376

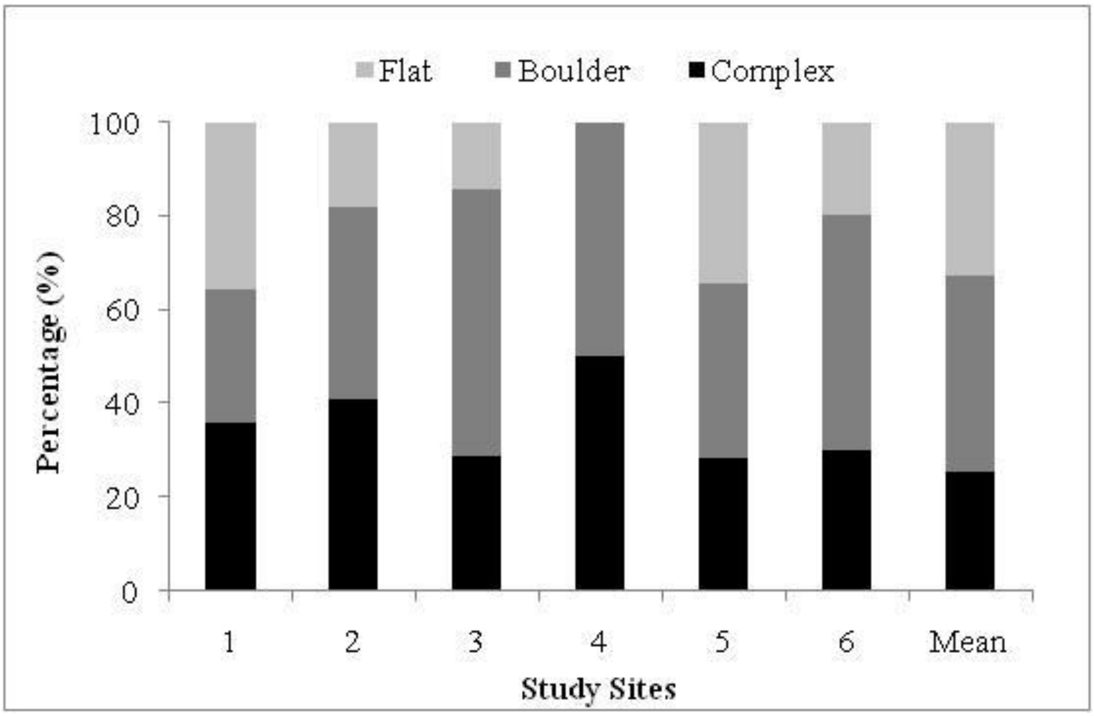

Figure 2.3. Percentage composition of numbers of 2 x 20 m transects per type of habitat at each study site in Palawan, Philippines.

In TRNP, seven permanent monitoring stations established in 2006 (Dolorosa et al. 2010) were surveyed and 7 more stations were added during the study. However, finding the exact location of the seven permanent monitoring stations was difficult because the concrete markers were missing and sometimes the GPS was pointing into the deep sea instead of the reef flat. In such instance, the site of the seven previously established permanent monitoring stations were approximately identified.

In Sites 5 and 6 (Rasa Island and Binduyan), nine and two stations were established respectively. During the survey, only reef areas with trochus or habitats suitable for trochus that are easy to assess were selected. For example, complex habitats thickly covered with branching corals which does not support trochus population were not selected, but rather a gradually sloping reef area composed of a mixture of different coral formations. Flat habitats dominated with rubble were selected and not those areas with thick cover of macro algae and patches of live corals. Transect surveys are a commonly used method to estimate the abundance of trochus, other invertebrates (Nash 1985; White 1986; Nash et al. 1995; Colquhoun 2001) and fishes (Hodgson et al. 2004).

Catch per unit effort (CPUE) of *Trochus niloticus* in unexploited and exploited sites

Catch per unit effort (CPUE), often expressed as volume of fish collected per unit of time, is a widely used measure of abundance in evaluating the status of fisheries (Bannerot and Austin 1983; Amar et al. 1996; Del Norte-Campos et al. 2003; Kaunda-Arara and Rose 2004). In this study, CPUE refers to the number of trochus captured per fisherman per hour (ind h^{-1}). At TRNP (Sites 1, 2 and 3), the estimated area covered during the collection was determined from the distance covered using the GPS coordinates and the approximate width (5 m) of the reef each person can survey visually while snorkelling. Collection was done when the water was calm and still as a precautionary measure and to minimise the effect of water current and waves on the numbers of collected shells. At Site 5, CPUE was based on the catches of 5 – 9 contracted octopus fishermen who showed up during a 4-day collection. Some fishermen did not show up after the first day as they find it hard to collect the trochus by breath hold diving at the deeper parts of the reef. At Site 6, CPUE was determined from the numbers of trochus collected by five fishermen spending a day fishing in Green Island Bay (also an area with several village MPAs). In these cases, the CPUE is based on fishing time and the number of trochus collected by each fisherman in Site 5, or all trochus collected by all fishers in Site 6. The areas covered by fishermen in Sites 5 and 6 were not known.

Population estimation of *Trochus niloticus*

A population estimate of trochus in complex habitat of Site 5 (Rasa Island) was determined from May 25 – 28, 2010 using a mark-recapture method assuming a closed population (Nash et al. 1995; Donovan and Welden 2002; Ogle 2010b). Trochus were collected by skin diving at the deeper parts of the reef by eight fishermen together with the main researcher for one day. They were inscribed with pencil marking on their apertures before releasing the shells at random locations in their natural habitat. Octopus collectors were again asked to collect shells in the succeeding three days or nights. During the succeeding days, the number of octopus collectors that participated in the recapture dropped by 50 % because some fishers find it hard to search for trochus by skin diving. Any unmarked shells collected on each day during the recapture were marked with distinct codes to distinguish the shells when recaptured in the succeeding days. A total of 70 trochus were marked, 6 of which were recaptured. A total of 94 shells (marked + unmarked) were captured in three days. None of the marked shells was recaptured twice. All the shells were released in their natural habitat after counting and inspection for any marking.

Temporal abundance of *Trochus niloticus* in TRNP

The abundance and sizes of trochus from seven monitoring stations at TRNP during the years 2006, 2008, 2009 plus the sizes of confiscated trochus in 2007 (see Dolorosa et al. 2010; Jontilla et al. 2011; Jontilla et al. in press) were graphically presented and compared with the sizes of trochus obtained during the recent survey. The length and width of transects during each year of survey, depth of surveyed habitat and survey methods at permanent monitoring stations are reflected in **Table 2.3**.

Table 2.3. Variation in length of transect lines and area covered during the survey at seven permanent monitoring stations in TRNP from 2006 – 2010. *In 2010, there were eight transects in six stations and 16 transects in one station.

Year	Width and length of transect per site	Number of transects per site	Area covered (m^2) per site	Total Area Surveyed (m^2)	Depth of surveyed area
2006	2 x 150	1	300	2 100	Shallow subtidal area
2008	2 x 100	1	200	1 400	Shallow subtidal area
2009	2 x 25	4	100	700	Shallow subtidal area
2010	2 x 20	8 or 16*	320 or 640	2 560	The boulder and complex habitats at shallow subtidal area

Data analyses

Exploratory analyses were conducted by comparing the substrate composition at each type of habitat. We found that one station at Site 1 (unexploited site) was devoid of trochus, possibly due to heavy sand deposits, so this was removed in the analyses. Further, we plotted the abundance of trochus with percentage of sand and found very few or absence of trochus in habitats with more than 10 % sand, so these transects were also deselected during the analyses. Analysis of variance was used to compare the substrate composition of each habitat type per site. We used the following dependent variables: rugosity index, hard corals, rock, rubble and sand with type of habitat (flat, boulder, complex) as fixed independent variables. We used Scheffé posthoc tests because of the uneven numbers of line transect per site. After we removed the transect lines with > 10 % sand, we got the following results: The amount of sand did not significantly vary among habitats ($F_{(2,312)}$ = 1.7, P = 0.184); Rugosity index significantly differed between habitats ($F_{(2,312)}$ = 315.42, P = < 0.001), as did for hard coral ($F_{(2,312)}$ = 56.15; P = < 0.001); rock ($F_{(2,312)}$ = 112.77, P = < 0.001), and rubble ($F_{(2,312)}$ =185.47, P = < 0.001).We found that the highest Rugosity index (1.22) and percentage of hard coral (20.5 %) occurred in complex habitat, the highest percentage of rock (82.22 %) was in boulder habitat, and the highest percentage of rubble (58.36 %) was in the flat habitat. The percentage of each type of substrate per type of habitat is presented in **Figure 2.4**.

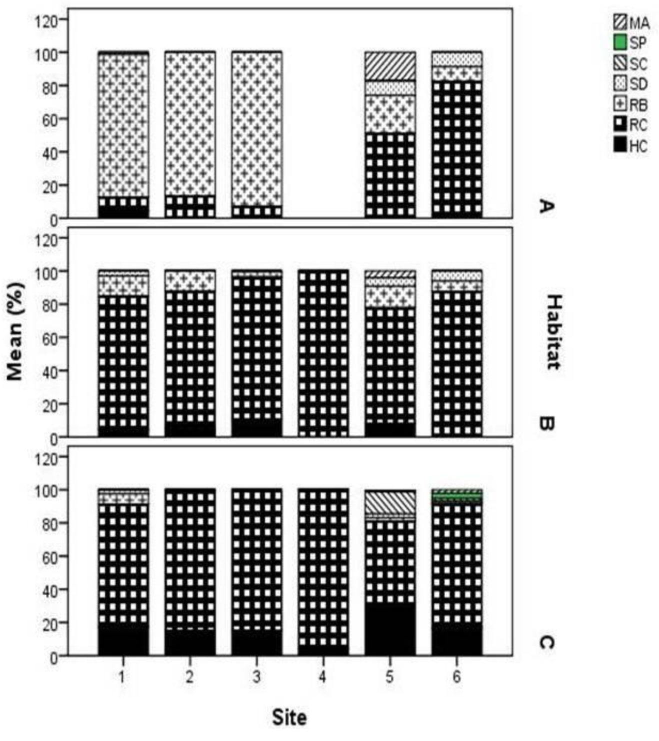

Figure 2.4. Average percent composition of substrate per type of habitat (A – flat, B – boulder and C – complex) in six study sites in Palawan, Philippines after deselecting transects with > 10 % sand. SP – sponge; MA – macro algae and seagrass but does not include the brown algae *Sargassum* spp.; RB – rubble; RC – rock; SC – soft coral and HC – hard coral.

Data on abundance were firstly explored in terms of differences between habitats and the effect of substrate type. Trochus does not occur in areas with a high percentage of sand, so we removed from the analysis the same transects mentioned earlier. We compared the abundance of trochus in three types of habitat within each site and among all sites. We used a Poisson log linear model to compare the abundance as this is appropriate for count data (Field 2009). In the analysis, we used the abundance (ind $40m^{-2}$) of trochus as dependent variable, with habitat or both habitat and status (Unexploited Site or Exploited Site) as factors. Rugosity index and percentages of rock and rubble were added as covariates.

A population estimate for Site 5 was determined using the Petersen method and its modified versions, the Chapman and Bailey models, and the Ricker model, a modification of the Chapman modification, all designed for a single census mark-recapture. This analysis makes all the following assumptions met: the population was constant; all shells have equal probability of being caught; tagging has no effect on their catchability; tags were not lost; and all captured trochus with tags were recorded (see Ogle 2010b). We used the FSA package (Ogle 2010a) for R software version R2.12.0 developed by the R Development Core Team (2011) to run the analyses.

In the analysis of the size structure, nearly 300 (21 % of the total) additional samples of trochus collected by octopus fishermen from the complex habitats at Sites 5 and 6 were included because only two trochus occurred along the transect line at the complex habitats at each site. In total, 1 446 individuals were measured, of which 6 % were from flat habitats, 40 % from boulder habitats and 54 % from complex habitats. More than 70 % of the samples were from Sites 1 and 2 **(Table 2.4).**

Table 2.4. Number of trochus sampled per site per type of habitat in six study sites in Palawan, Philippines. A – flat, B – boulder, C – complex, UE – unexploited, ME – moderately exploited, OE - overexploited * collected by octopus fishers for 1 – 2 days, NT – no transect.

Site Number	Site Name	Status	Habitat Type			Total
			A	B	C	
1	North Atoll, TRNP	UE	20	312	285	617
2	North Atoll, TRNP	ME	58	209	162	429
3	South Atoll, TRNP	OE	NT	26	46	72
4	Jessie (Beazley Reef, TRNP	OE	NT	0	0	0
5	Rasa Island, Narra, Palawan	OE	11	14	155*	180
6	Binduyan, Puerto Princesa City & part of Green Island Bay, Palawan	OE	5	16	127*	148
	Total		94	577	775	1 446

The mean sizes of trochus were compared according to type of habitat across sites and between habitats within each site. We used univariate analysis of variance to compare the sizes of trochus in each habitat across sites or sizes of trochus across habitats for each site. In such case, the sizes of trochus were treated as dependent variable and either site or habitat as independent variable. Post hoc multiple comparisons of means were conducted using Scheffé test as there were unequal numbers of samples per type of habitat or site. Comparisons were conducted using SPSS version 16 (Field 2009).

The sizes of trochus from the seven permanent monitoring stations at TRNP obtained in 2006, 2008 and 2009 (see Dolorosa et al. 2010; Jontilla et al. 2010; Jontilla et al. in press) and during the recent survey were compared using univariate analysis of variance. In 2007, there was no survey of the site so the sizes of trochus confiscated from illegal fishermen at TRNP in 2007 (see Dolorosa et al. 2010) were included in the analysis. In the analysis, the sizes of trochus were treated as dependent variable and the year of sampling as independent variable. The analysis was performed using SPSS version 16 (Field 2009).

RESULTS

Abundance of *Trochus niloticus* in unexploited and exploited habitats

The overall pattern of abundance at six study sites was characterised by a declining trend with increasing site's distance from the Ranger Station as for the case of study sites at TRNP, with loss of large trochus from the flat habitat first, then from boulder habitats, but with large individuals remaining in complex habitats even in sites that are relatively far from the Ranger Station or the sites that are close to coastal communities. The densities of trochus in boulder and complex habitats at Site 1 were considerably higher than at other sites (**Figure 2.5**). In the boulder habitat at Site 1, the abundance was nearly twice as high as that in the complex habitat, but the difference was not so pronounced in Site 2. In Site 3, the trend changed, with abundance in the complex habitat higher than in the boulder habitat. No trochus were found at Site 4, while extremely low trochus abundances occurred in all habitats at Sites 5 and 6.

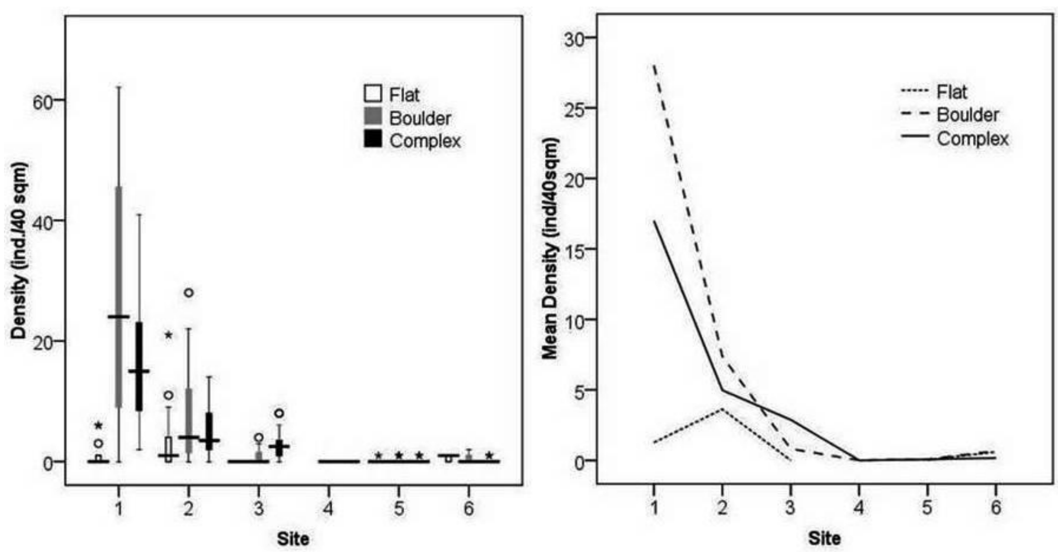

Figure 2.5. Box plot (left) and line graph (right) showing the density of trochus per type of habitat per site. Site 1 was unexploited and Sites 2 – 6 were exploited at varying degrees.

In general, the abundances of trochus in three types of habitat were more variable and significantly higher in the unexploited site than at the exploited sites. The boulder habitat held the highest trochus abundance at the unexploited site, while in heavily exploited sites, trochus were abundant at complex habitats. Comparison of abundance at each type of habitat per site revealed a significant variation in abundance at Site 1 ($\chi^2 = 11.20$, $df = 2$, $P < 0.01$), but not at Site 2 ($\chi^2 = 0.93$, $df = 2$, $P > 0.05$), Site 3 ($\chi^2 = 0.64$, $df = 3$, $P > 0.05$), Site 5 ($\chi^2 = 51$, $df = 2$, $P > 0.05$) and Site 6 ($\chi^2 = 0.35$, $df = 2$, $P > 0.05$). Multiple comparison of all sites revealed significant variation in abundance among sites ($\chi^2 = 143.08$, $df = 4$, $P < 0.001$, **Table 2.5**).

At Site 1, the abundance of trochus varied significantly in association with habitat, percent rock and rubble, the interaction between habitat and percent rock, and the covariates such as rock and rubble. At Site 2, the abundance varied significantly in association with rugosity index and on the interaction between habitat and rugosity index. At Site 3, Site 5 and Site 6, no significant variation was noted on abundance in association with habitats, covariates and the interaction between habitats and covariates. Comparison of all Sites revealed that abundance of trochus varied significantly in association with habitat, covariates and the interaction between habitats and covariates (**Table 2.5**).

Table 2.5. Wald χ^2 and Probability values obtained using a Generalized Linear Model (Poisson log linear model) indicating the extent of effects of different variables on trochus abundance at each site or their combinations. S – site, RI – rugosity index, RC – rock, RB – rubble. Shaded values are significant.

		S1		S2**		S3**		S5		S6		Multiple Comparison of Sites			
Source	**df**	**Wald χ^2**	**Sig.**	**Wald χ^2**	**Sig.**	**Wald χ^2**	**Sig.**	**Wald χ^2**	**Sig.**	**Wald χ^2**	**Sig.**	**Source**	**df**	**Wald χ^2**	**Sig.**
(Intercept)	1	0.71	0.400	0.47	0.491	0.14	0.710	2.88	0.089	0.20	0.658	(Intercept)	1	13.24	<0.001
Habitat	2	11.20	0.004	0.93	0.336	0.64	0.422	0.51	0.774	0.35	0.840	Habitat	2	37.07	<0.001
Site												Site	4	143.08	<0.001
RI	1	0.07	0.796	8.54	0.003	0.06	0.805	0.02	0.900	0.58	0.447	RI	1	11.01	0.001
RC	1	17.67	<0.001	0.17	0.681	2.10	0.148	1.23	0.267	0.09	0.769	RC	1	9.28	0.002
RB	1	14.63	<0.001	0.51	0.477	2.20	0.138	0.87	0.350	0.11	0.744	RB	1	9.92	0.002
Habitat * Site												Habitat* Site	7	95.19	<0.001
Habitat * RI	2	3.73	0.155	4.41	0.036	0.69	0.406					Habitat*RI	2	45.24	0.038
Habitat * RC	2	15.66	<0.001	0.19	0.664	0.18	0.668					Habitat* RC	2	6.53	<0.001
Habitat * RB	2	4.19	0.123	2.04	0.153							Habitat* RB	2	32.14	<0.001

* interaction

**all df = 1, because flat habitats were removed due to extremely high deviance (> 500); or some convergence criteria were not satisfied

Highlighted values are significant

Plotting the abundance of trochus against rugosity index clearly indicates the absence of proportional relationship. A high abundance of trochus occurred only when rugosity index at unexploited and moderately exploited sites were between 1.05 and 1.2, but this pattern is no longer clear in heavily exploited sites such as Sites 3 – 6 (**Figure 2.6**). The percentage of rocks tended to positively influence the abundance of trochus in boulder and complex habitats which is clearly visible in unexploited site (Site 1). This pattern began to disappear in exploited sites (**Figure 2.7**). By contrast, the abundance of trochus is inversely proportional with percentage of rubble (**Figure 2.8**).

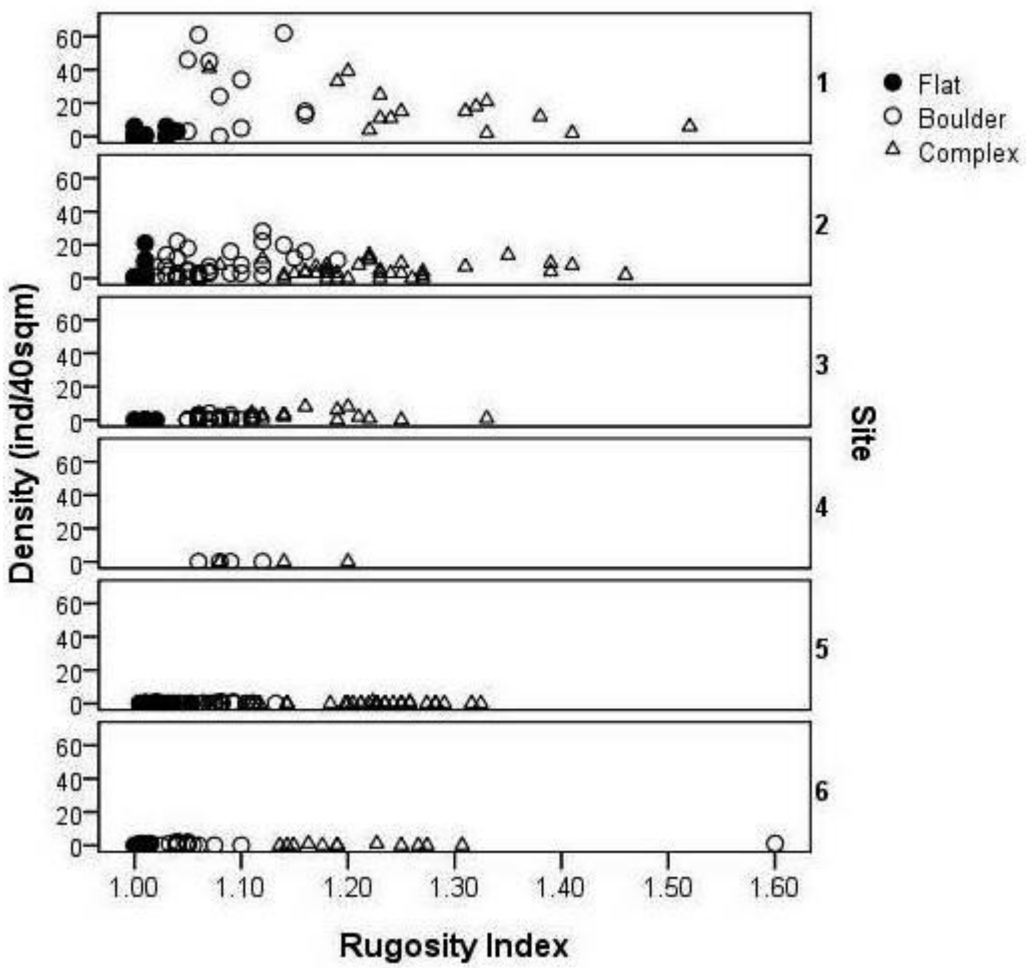

Figure 2.6. Relationship between trochus abundance and rugosity index in three types of habitat at six study sites in Palawan, Philippines.

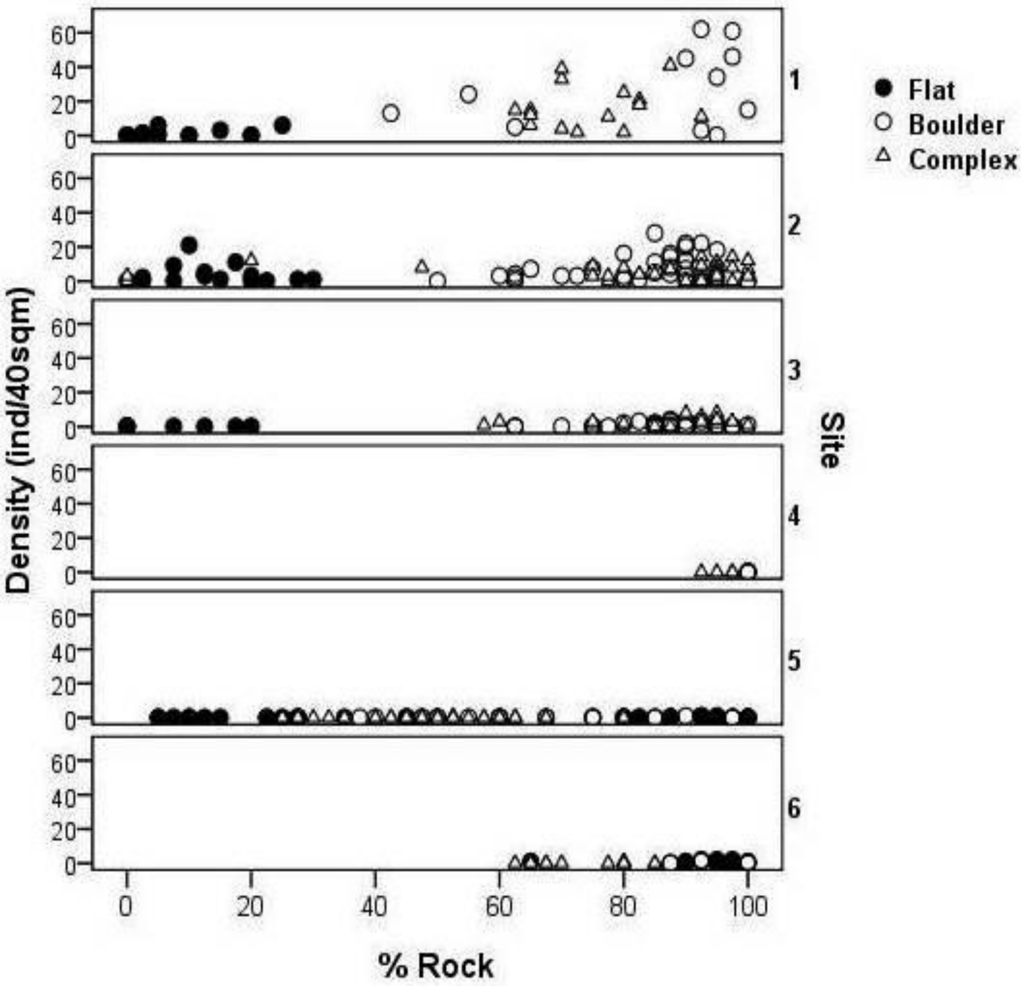

Figure 2.7. Relationship between trochus abundance and percentage of rock in three types of habitat at six study sites in Palawan, Philippines.

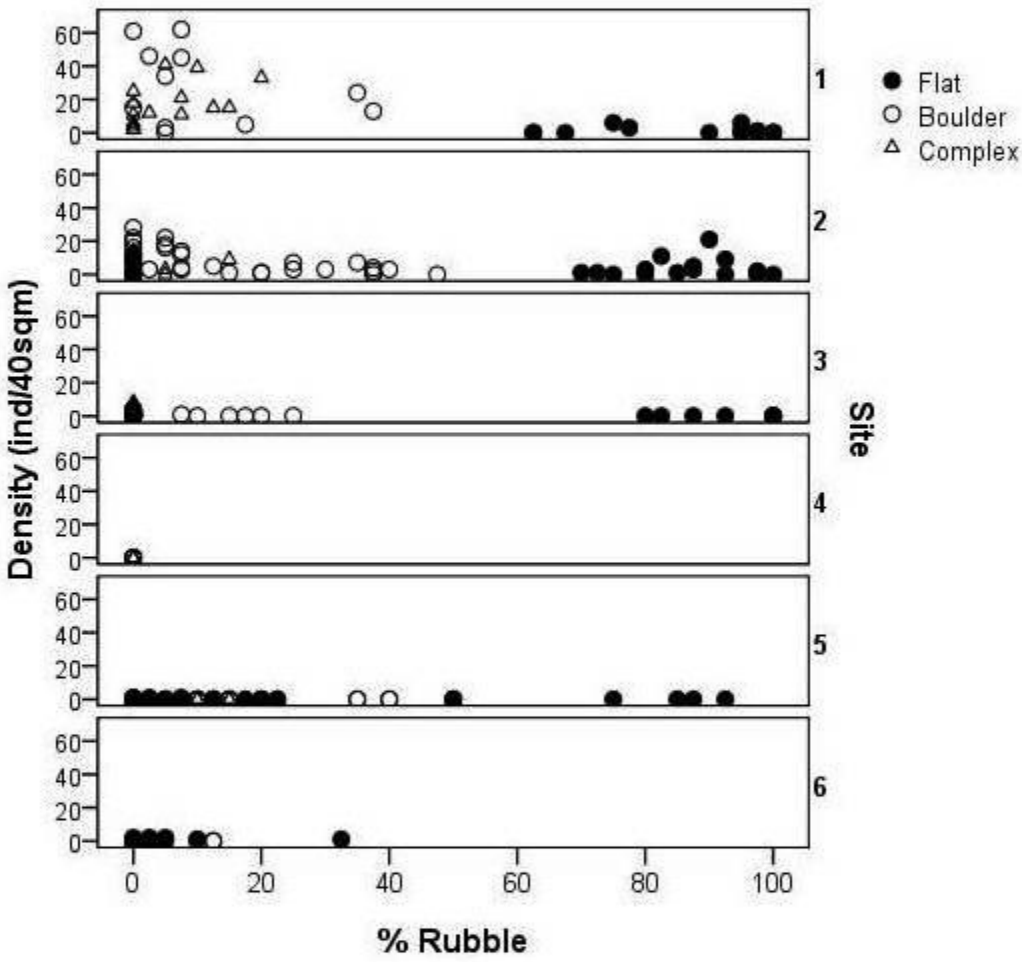

Figure 2.8. Relationship between trochus abundance and percentage of rubble in three types of habitat at six study sites in Palawan, Philippines.

The density of trochus and CPUE was much higher in unexploited and moderately exploited sites compared to heavily exploited sites. At Site 5 and Site 6, catches were extremely low, with each fisherman only able to collect about 1 – 2 trochus per hour, compared with the high CPUE of nearly 200 ind hr^{-1} at unexploited site. The area covered during the collection increased as the density and CPUE of trochus decreases. The large confidence intervals recorded in unexploited (Site 1) and moderately exploited site (Site 2) suggested a high variation in abundance within the complex habitats. At Site 4, only empty shells of trochus were found, indicating that they have occurred in the past on the reef (**Figure 2.9**).

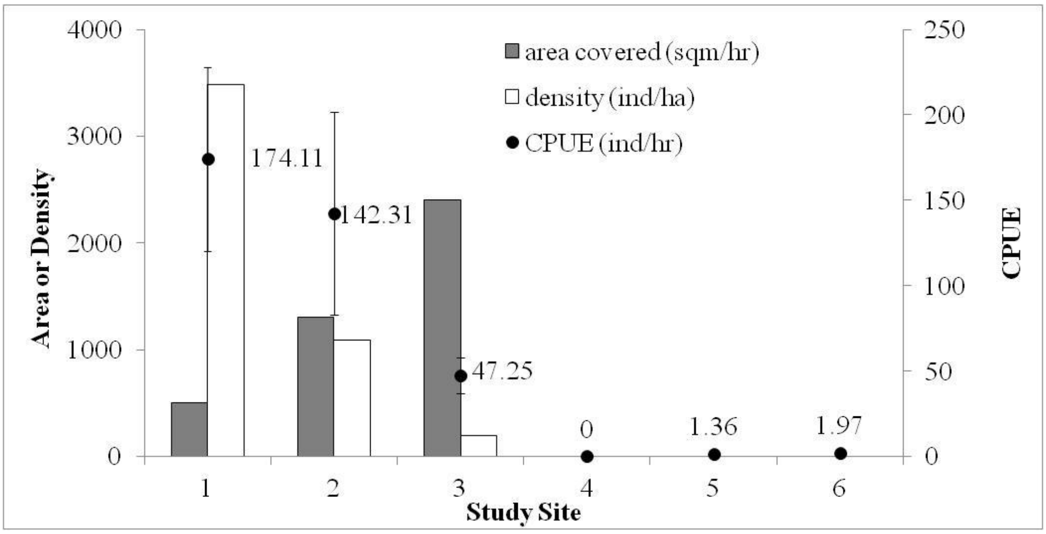

Figure 2.9. CPUE for trochus in complex habitats of six study sites in Palawan, Philippines. Error bars are 95 % confidence intervals. Area covered (m^2 hr^{-1}) during the collection and estimated density (ind ha^{-1}) were only for Sites 1 – 3.

The estimated population of trochus in the complex habitat of Rasa Island was about 1 000 individuals with 95 % confidence intervals from 500 to either 2 000 or 3 000 individuals depending upon the statistical model used. Although the estimated population obtained using the Petersen model has an inflated upper confidence interval which is possibly a characteristic of the model, estimates of population using the other two modifications of Petersen model, the Chapman and Bailey models and the modification of Chapman modification, the Ricker model, have relatively similar population estimates and reduced upper confidence intervals compared with Petersen model (**Figure 2.10**). Assuming that the complex habitat was about 60 ha (Widmann et al. 2010), this would result to an estimated density of 17 ind ha^{-1}, comparable to the estimated value of 15 ind ha^{-1} (or 0.06 ind 40 m^{-2}) obtained by the transect survey. However, using the Palawan Council for Sustainable Development (PCSD) map gives a coral reef area of approximately 27 ha, which would give a density of 40 ind ha^{-1}.

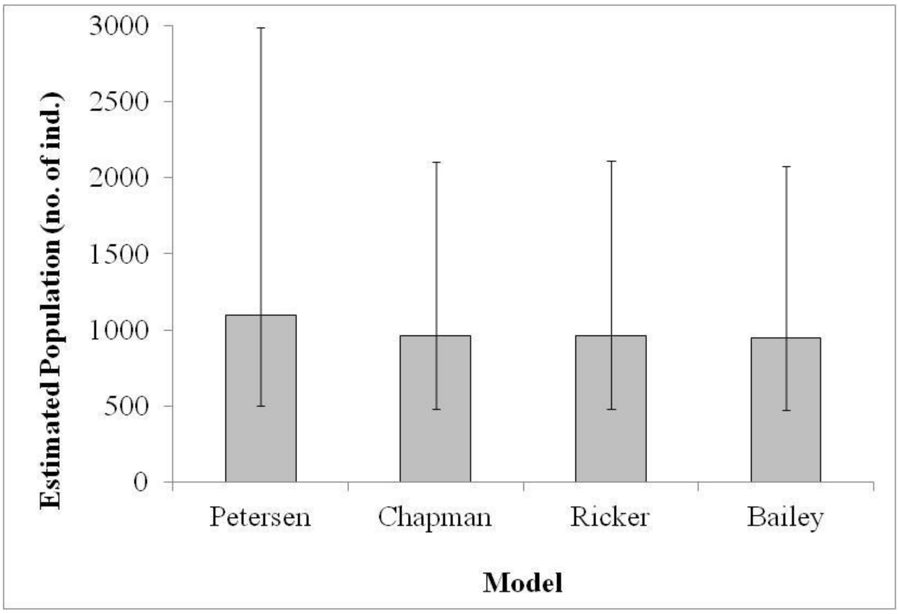

Figure 2.10. Population estimates of trochus at Site 5 (Rasa Island) obtained using the Petersen model and its modifications, the Chapman and Bailey models, and the modification of Chapman modification, the Ricker Model. Error bars are 95 % confidence intervals.

Size structure of *Trochus niloticus* in unexploited and exploited habitats

The size structure of trochus varied at each habitat with a general declining trend in sizes from unexploited to heavily exploited sites (**Figure 2.11, Table 2.6**). All sizes (small and large) of trochus occurred on flat habitats at unexploited site (Site 1, mean size: 87.47 mm), while only small sized individuals occurred on flat habitats of exploited sites (mean sizes ranged between 30.80 - 45.82 mm). Large individuals (mean: 87.39 mm) were common on boulder habitats at unexploited site (Site 1) and moderately exploited site (Site 2, mean: 95.85 mm), but were absent on boulder habitats at heavily exploited Sites 5 and 6 as indicated by sampled trochus having an average size of 48.58 and 42.50 mm respectively. Large individuals occurred in all complex habitats except at site 4. In Sites 5 and 6, only two or three individuals occurred along the transect lines at complex habitat, so the catches of contracted octopus fishermen for population estimation (**this Chapter**) and translocation (**Chapter 6**) were included in the graph and analysis (**Table 2.6**).

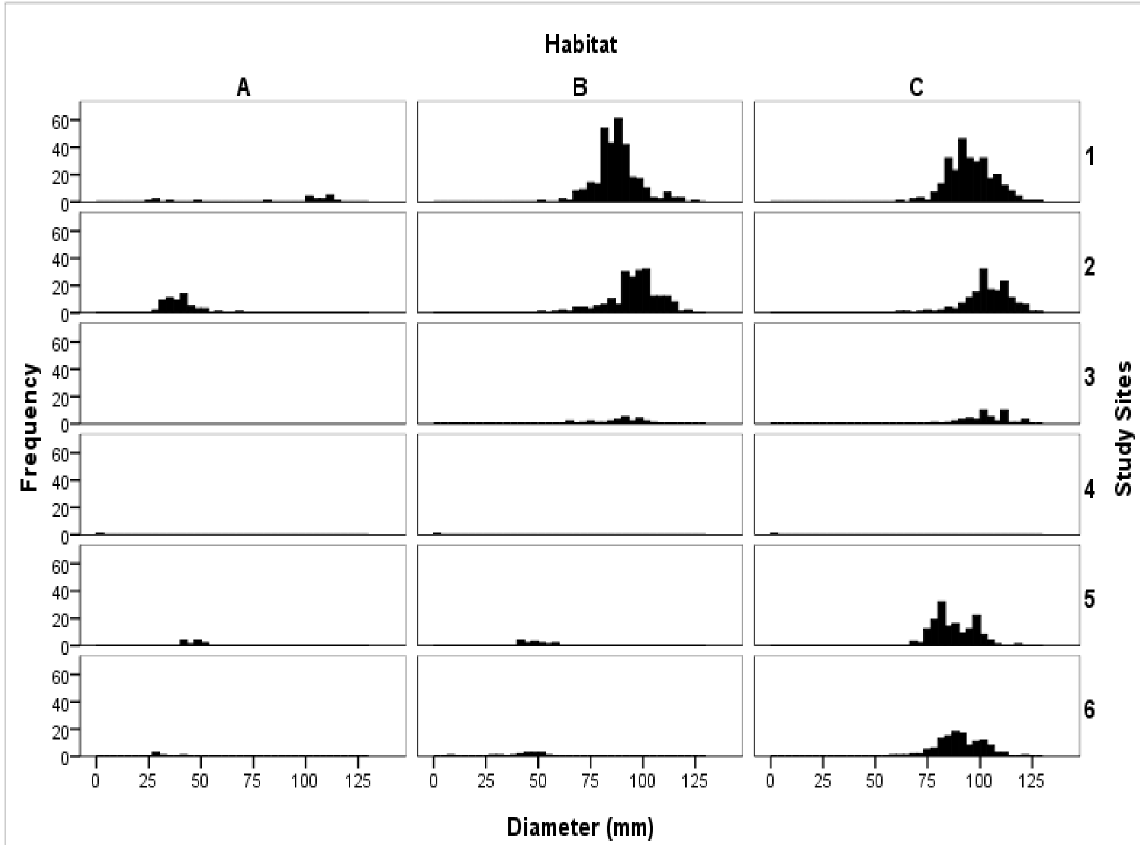

Figure 2.11. Frequency histogram of trochus per type of habitat at six study sites. A – flat habitat, B – boulder habitat, C – complex habitat. Trochus in complex habitat of Site 5 and Site 6 were from octopus fishermen.

Comparing the sizes of trochus per habitat across sites using analysis of variance revealed that trochus on flat habitats at the unexploited site (Site 1) were significantly larger than at moderately or overly exploited sites ($F_{(3,90)}$ = 44.74, P = < 0.001). In boulder habitats, the sizes of trochus were not significantly different among Sites 1, 2 and 3, but these were significantly larger than in Sites 5 and 6 ($F_{(4,572)}$ = 141.74, P = < 0.001). However, in the complex habitat at the unexploited site, trochus was significantly smaller than in Sites 2 and 3 but not in Sites 5 and 6 ($F_{(4,770)}$ = 59.11, P = < 0.001). Comparing sizes by habitat within each site revealed that in Site 1, the average size of trochus in flat and boulder habitats were not significantly different but both were significantly smaller than in complex habitat ($F_{(2,614)}$ = 36.91, P = < 0.001). Similar results were noted in Site 5 ($F_{(2,177)}$ = 204.43, P = < 0.001). In other sites, the largest average size occurred on the complex habitats, followed by those on the boulder and flat habitats. The overall mean sizes of trochus per type of habitat revealed that on complex habitats, trochus were significantly larger than those found on boulder or flat habitats ($F_{(2,1443)}$ = 403.75, P = < 0.001, **Table 2.6**).

Table 2.6. Mean diameter (mm) of trochus per habitat per site. A – flat, B – boulder, C – complex. ns – no shells found along the transect lines or within the habitat; nt – no transect line was laid. The same superscript means no significant difference.

Site Number	Habitat across Sites			Habitats within Site			df	F value	P value
	A	B	C	A	B	C			
1	87.47^{a}	87.39^{a}	95.55^{b}	87.47^{b}	87.39^{b}	95.55^{a}	2; 614	36.91	<0.001
2	39.41^{b}	95.85^{a}	103.28^{a}	39.41^{c}	95.85^{b}	103.28^{a}	2; 246	735.58	<0.001
3	ns	88.19^{a}	102.93^{a}	ns	88.19^{b}	102.93^{a}	1; 70	35.37	<0.001
4	nt	ns	ns	nt	ns	ns			
5	45.82^{b}	48.58^{b}	87.35^{c}	45.82^{b}	48.58^{b}	87.35^{a}	2; 177	204.43	<0.001
6	30.80^{b}	42.50^{b}	90.28^{c}	30.80^{c}	42.50^{b}	90.28^{a}	2; 145	192.84	<0.001
Mean				49.93^{c}	88.30^{b}	95.10^{a}	2; 1443	403.75	<0.001
Df	3; 90	4; 572	4; 770						
F value	44.74	141.74	59.11						
P value	<0.001	<0.001	<0.001						

Compiled data on abundance of trochus at the permanent monitoring stations in TRNP from 2006 (Dolorosa et al. 2010; Jontilla et al. 2010, Jontilla et al. in press) up to the present showed a massive decline at stations 3 – 4 (part of moderately exploited North Atoll) and stations 5 – 7 (part of heavily exploited South Atoll). Variations in yearly abundance occurred in stations 1 – 2 (part of unexploited North Atoll) over a period of four years but were relatively stable and much higher compared to the abundances at exploited stations (**Figure 2.12**).

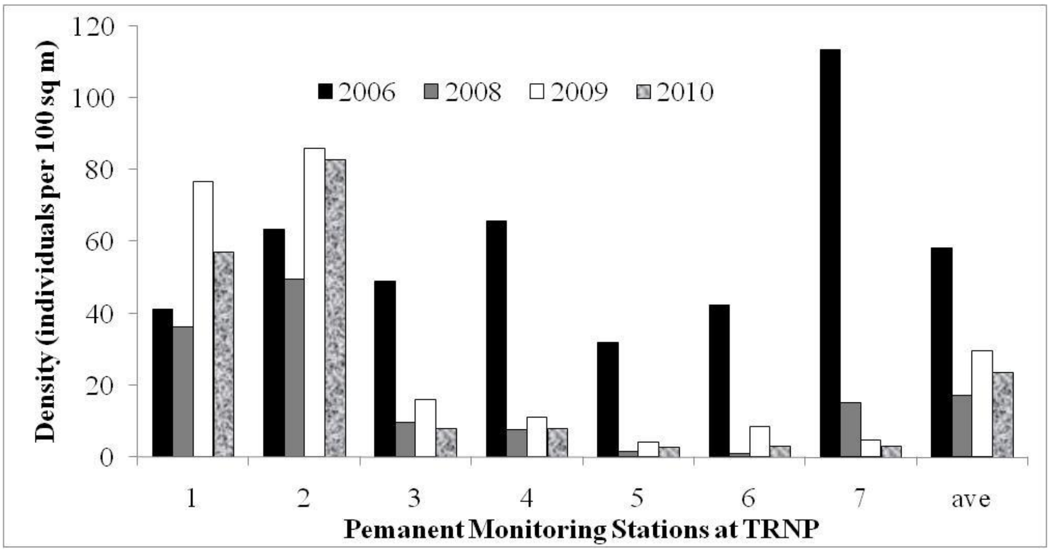

Figure 2.12. Trend in abundance of trochus at permanent monitoring stations in TRNP from 2006 – 2010. Number of samples: 2006 (n=1217), 2008 (n=240), 2009 (n=413), 2010 (n=509).

There was a gradual increase in the average mean sizes of trochus from 2006 to 2010 (**Figure 2.13**). Univariate analysis of variance suggested a significant difference among the mean sizes of trochus at seven permanent monitoring stations at TRNP from 2006 – 2010 ($F_{(4, 3036)}$ = 442.27; P = < 0.001). Post hoc Scheffé tests showed that the sizes of trochus in 2006 (66.96 mm ± 17.39 sd) were not significantly different with the mean size of confiscated trochus in 2007 (68.51 mm ± 9.84 sd). In 2008, the mean size of trochus (82.34 mm ± 13.81 sd) was significantly larger than in 2006 and 2007 but significantly smaller than in 2009 (93.50 ± 15.31) and 2010 (89.38 ± 10.52). Trochus sampled in 2009 were significantly larger than in 2010.

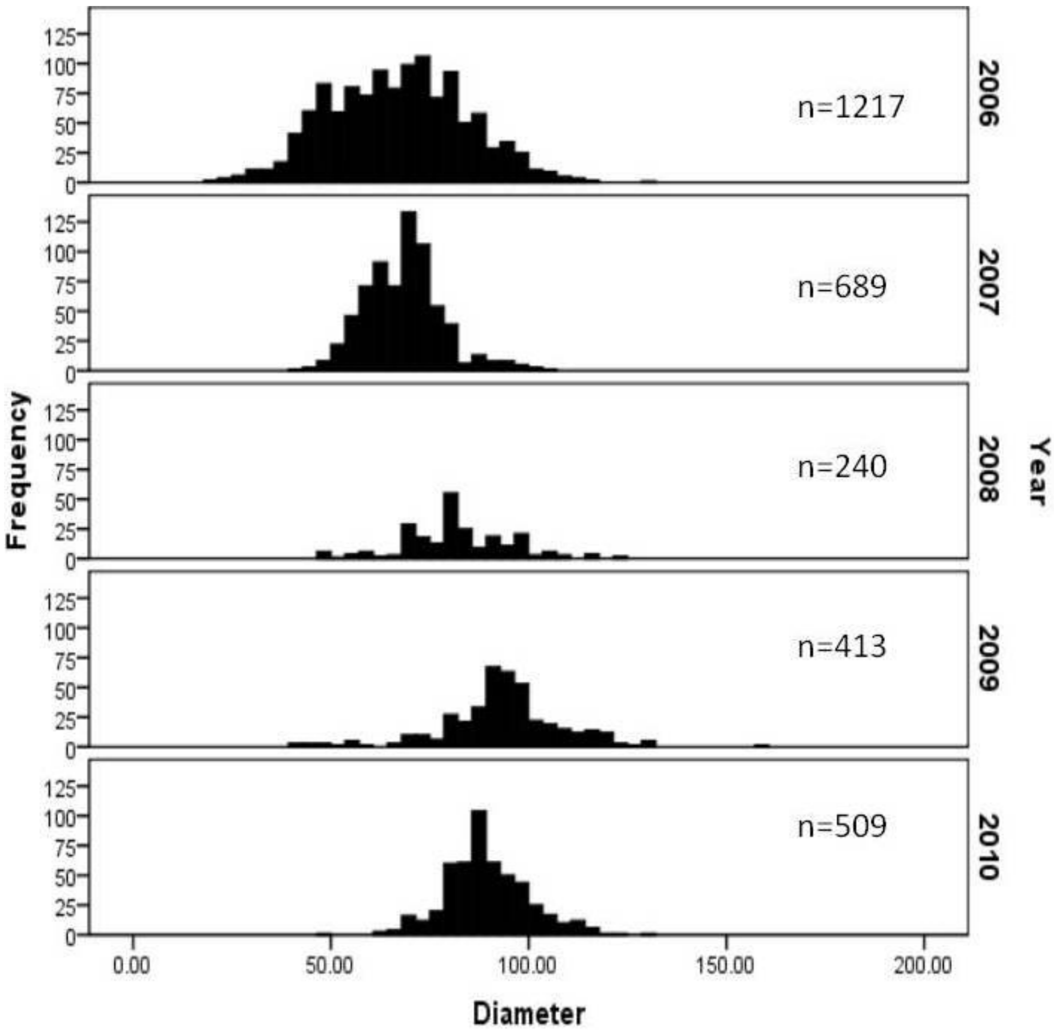

Figure 2.13. Frequency histogram of trochus at seven permanent monitoring stations at TRNP from 2006 – 2010. The data in 2007 were the sizes of confiscated trochus from illegal fishermen at the park.

DISCUSSION

The pattern of abundance and size structure of trochus at the unexploited site is similar to that in other studies (Sims 1984; Nash 1985; Smith 1987; Castell 1993; Nash 1993; Castell 1997; Colquhoun 2001), where large trochus occur in a wide range of depths. But exploitation can significantly reduce their abundances (Burhanuddin 1997; Smith et al. 2002). The greatest abundance of adults occurred at the intertidal area with decreasing abundance towards the deep (Nash 1985; Nash 1993), while the juveniles are restricted to intertidal flats (Castell 1997; Colquhoun 2001).

It is not clearly understood why juveniles only occur in intertidal areas. This has been linked with the timing of spawning during spring tides where the eggs are carried to the highest part of the reef (Nash 1993). It has also been attributed to high predation on individuals that settled on subtidal habitats (Nash 1993), or in difficulty in finding them because of their size, cryptic nature (Castell et al. 1996) and habitat complexity. A study on recruitment abundance of three species of kelp forest trochid gastropods (*Tegula* spp) which closely coincided with adult distribution suggests that settlement occurs at different depths (Watanabe 1984). Trochus juveniles raised in cages set on deeper parts of the reef (**Chapter 5**) can grow as fast as those at the intertidal area (**Chapter 6**), suggesting that conditions in deeper parts of the reef are suitable for growth.

Predation can highly influence the population size and size structure of other intertidal organisms. For example caging experiments on the limpet *Patella vulgata,* showed that predation reduced limpet abundance by 50 % in two months, with the heaviest mortality of smaller individuals on uncaged and partly caged treatments compared to complete cage treatment (Silva et al. 2008). Predation can also determine the lower limit of other intertidal species. The removal of all predatory starfish (*Pisaster ochraceus*) from a stretch of shore allowed the barnacle *Balanus glandula* and mussel *Mytilus californianus* to survive on lower parts of the shore, than in an adjacent control area (Connell 1972). Caging experiments on three species of barnacles protected the species from predation at various shore levels (Connell 1972 and references therein) leading to a substantial increase in density. In intertidal areas, highly mobile predators are active when the tide is in (Silva et al. 2008; pers. obs.), suggesting that level of predation is higher in constantly submerged habitat than at periodically exposed habitats. Trochus are preyed upon by some vertebrates such as fishes, marine turtles and a variety invertebrates species (Nash 1993; Isa et al.

1997). The hermit crabs are important cause of mortality on adult trochus (Nash 1985). As predation on intertidal organisms is higher during high tide (Silva et al. 2008), while the removal of predators has increased the survival of prey at subtidal habitats (Connell 1972), it is likely that predation and habitat complexity have a significant influence on the size structure and abundance of trochus in unexploited habitats.

Other possible reasons affecting the pattern of size and abundance at different depths could be habitat complexity and aggregation tendency of trochus. In his survey, Colquhoun (2001) noted that juvenile trochus occurred only on the rock habitat if it was bordered by a platform or patch reef habitat, but not if bordered with sand. He also noted that trochus occurred in a very distinct narrow zone within the intertidal rock habitat which provides more protection from desiccation, water currents, and predation compared with the rubble habitat. Castell (1997) also found that sizes of trochus juveniles increased from small rubble, to large rocks and coral bench, and from shallow pools to deeper pools. This is possibly because rocks and coral bench can provide better refuge than rubble, and could explain the high abundance of trochus in the boulder habitat at the unexploited site.

The importance of habitat in enhancing survival has been reported for other species. For example, the association of juvenile abalone with crustose coralline algae is believed to be important for food and as a refuge from predators like wrasses which prey only on abalone juveniles larger than 5 mm long (Shepherd and Turner 1985). Ray-Culp et al. (1999) on the other hand found that although the increases in seagrass refuge did not affect the levels of predation by crabs on juvenile queen conch *Strombus gigas*, the proportional mortality due to predation was higher on smaller individuals, and mortalities declined with prey densities. This suggests that aggregation can help reduce the impact of predation, because a group of large shells could be a more effective juvenile refuge than seagrass blades. In our survey at flat habitats at TRNP, trochus juveniles occurred only under small rocks scattered on rubble dominated habitat. This suggests that boulders, like the aggregating adults, can provide safe refuge against predators and against extreme heat during day low tide. In all surveyed boulder habitats, although we did not find a high density of juveniles, in Site 1, we found a dense aggregation (of least 3 – 5 individuals per group) of medium sized (70 – 90 mm) adults appearing to be stunted because of their worn out shells and flanged aperture. This was also the site with the highest abundance of nearly 2 ind m^{-2} in two 2 x 20 m transects. Complex habitats although provide intricate crevices suited for trochus, this is constantly underwater, having a high predation pressure than on boulder habitat at the intertidal area, the possible reason why fewer trochus are found.

Other factors affecting the variation of density and size distribution between habitats and sites could include: patterns of and restricted larval dispersal, water currents, habitat size and suitability, growth density dependence and degree of exploitation. Trochus have a short larval period, settling 50 – 60 h after fertilisation although some larvae could remained planktonic up to 10 days in the absence of suitable substrate (Heslinga 1981; Heslinga and Hillmann 1981; Nash 1993), thus having a restricted dispersal (Nash 1993). It is not known how far a trochus larva can travel before settling on the reef, but because of their short larval period (Heslinga 1981), patchy distribution (Carpenter and Niem 1998) and ability to colonise offshore reef areas when translocated (Smith 1987; Gillett 1993; Ponia et al. 1997; Zoutendyk 1997) suggest that recruitment is within the parent population (Nash 1993). For greenlip abalone *Haliotis laevigata,* studies revealed that recruitment was independent of adult density at 100 m segments but were influenced by the topographic features of the coast. Depending on water current, the larvae could be transported at a distance of no less than hundreds of meters and the larvae are concentrated on sites where there are eddies and retention zones (Shepherd et al. 1992).

The highest adult density of 15 000 ind ha^{-1} or an average of 7 000 ind ha^{-1} at boulder habitats of the unexploited site at TRNP, is similar to other reports on trochus' pattern of abundance (Nash 1993). The highest density of trochus was reported from Cook Islands (66 ind m^{-2} or 66 000 ind ha^{-1}) (see Tsutsui and Sigrah 1994). Variation in density can be a result of the tendency of trochus to aggregate especially when spawning (Nash 1993).

In overexploited sites, the abundances were extremely low and the patterns of distributions were altered depending on the level of exploitation, size and location of the reef, forcing fishermen to venture in offshore areas and reefs of adjacent countries. In heavily exploited Cartier Reef in Australia, no trochus were recorded at the reef flat, 1.33 ind ha^{-1} at the reef crest (2 – 8 m deep), and 3.33 ind ha^{-1} at the reef slope (9 – 24 m deep). Only sizes 55 – 125 mm were recorded and the absence of juveniles was attributed to their characteristics of being cryptic or because they occur in areas that are easily fished (Smith et al. 2002). In this study, the abundance of trochus at heavily exploited Rasa Island obtained using the information on estimated population and area of the reef of about 15 – 40 ind ha^{-1} was much higher than in Cartier Reef, but the average density obtained by transect survey at three types of habitats was much lower, ranging only from 0.5 – 1.0 ind ha^{-1}. In Binduyan (Site 6) the average density at boulder (7.5 ind ha^{-1}) and complex (3.5 ind ha^{-1}) habitats obtained by transect survey was a little higher than in Rasa Island, and the recruits at flat habitat occurred at 20 ind ha^{-1}. The frequency of fishing at off shore Jessie Beazley Reef (Site 4) could be less than at Sites 5 and 6, but no trochus were found. Because of its small size, the reef could have been easily overfished and recruitment coming from other reefs could be difficult because of its remoteness from other reefs areas. The surveyed sites in this study only covered up to about 3 – 5 m deep and do not include areas unreached by breath hold dive. Assessment of deeper reef areas in the mainland Palawan would shed more light on the status and potential of trochus for conservation. The low CPUEs at Sites 5 and 6 also support the low numbers of trochus encountered during the transect surveys. The low abundance in most parts of Palawan along with its high market price have forced fishermen to fish trochus in MPAs like TRNP (Dolorosa et al. 2010; Jontilla et al. in press) or even in Malaysian waters (pers. comm. with residents from southern Palawan).

The variation in sizes of trochus in exploited sites which were either smaller or larger than in the unexploited site could be an effect of different levels of harvesting and faster growth at reduced density. In Sites 2 and 3, reduction in density could have favoured the growth of the remaining shells as reflected by the larger sized trochus found in boulder habitat at Site 2 than in Site 1 (**Table 2.6, Figure 2.11**). Nash (1993) suggested that in the wild trochus growth increases with declining density, based on the observation of shell processors that trochus shells become thinner as years of harvesting progress. In Sites 5 and 6, although it is presumed that fast growth occurs because of low density, their sizes are significantly smaller compared to other sites possibly because most shells are harvested as soon as they emerge and assume a non cryptic behaviour. Octopus fishers interviewed during the survey are aware that the harvest of trochus is prohibited but trochus are collected once encountered while diving on the reef.

Some variation in juvenile shell morphology was noted during the survey, with those at TRNP (< 20 mm) having a smooth and solid shells while those in Binduyan (Site 6) of the same size or even larger (> 25 mm) were fragile, thin and with serrated edges. The thickness of trochus shells increases with size (Nash 1985) but the degree of increment might vary depending on the growth rate which is influenced by environmental factors and type of habitats. The report of Purcell (2002) that shells of wild trochus juveniles have serrated margin until they reach 25 – 30 mm agreed with our samples from Site 6, but not for samples from TRNP. The shift in shell morphology of wild juveniles occurs with behavioural shift from cryptic to non cryptic behaviour (Purcell 2002). However, during our day surveys, many sub-adults (<50 mm and without serrations) were still cryptic under rock crevices. Variation in shell formation has been reported for the intertidal limpet *Notoacmea scapha* (Nakano and Spencer 2007), and the intertidal snail *Nodilittorina australis* can switch between two morphologies when translocated (Yeap et al. 2001). The limpets *Lottia asmi* and *L. digitalis* can change their colour depending on the food they take in and shell formation changes in responses to the topographical complexity and constraints of substrata (Lindberg and Pearse 1990). Cryptic colouration caused by lichen on intertidal limpets could help reduce the impact of visual predators such as birds (Espoz et al. 1995).

Estimating the densities of juveniles is difficult because they occur in a varied types of habitat in the intertidal areas, and they are very cryptic, so a large number of juveniles could missed during the surveys (Castell et al. 1996; Castell 1997; Colquhoun 2001; Crowe et al. 2001). However, the low numbers of recorded juvenile densities (0 – 0.09 ind m^{-2}) in this study compared to a reported average density of 0.1789 and 0.115 ind m^{-2} at small rubbles, large rocks and coral bench habitats in Queensland, Australia (Castell 1997) could be related to the levels of exploitation and type of habitat surveyed and not necessarily due to the number of missed individuals. The surveyed flat habitats at TRNP were generally bare with few rocks where the juveniles were found hiding, while flat habitats at Rasa and Binduyan (generally composed of flat solid coral rocks) were frequently trampled with gleaners. According to Schiel and Taylor (1999) trampling on intertidal habitat can cause disturbance on many species.

The illegal exploitation of trochus at TRNP has greatly reduced its average density at the seven permanent monitoring stations of about 60 ind 100 m^{-2} in 2006 (Dolorosa et al. 2010) by 70 % in two years (Jontilla et al. in press). The overall mean abundance in 2010 (23.64 ind 100 m^{-2}) was a little higher than in 2008 (17.14 ind 100 m^{-2}) but lower than in 2009 (29.50 ind 100m^{-2}) (Jontilla et al. 2011; Jontilla et al. in press). This decline in abundance between 2009 and 2010 could still be a possible effect of continued illegal fishing at the Park because although we have increased the area covered with transect lines in 2010 (2 560 m^2) more than three times higher in 2009 (700 m^2), only 509 individuals were sampled in 2010 compared to 413 individuals in 2009 (**Table 2.3, Figure 2.13**). Although no trochus poachers were apprehended between 2009 and 2010, a group of shark fishermen were apprehended in October 2009 and during my stay at the park in December 2009, we spotted on a radar two moving bodies (possibly two small outrigger motor boats) near Malayan Wreck area, but during that time the patrol boat was under repair, and the waves were high that it would be too risky to check the area with a dinghy. If poaching remained uncontrolled at TRNP even at a lesser degree, trochus population especially in sites distantly located from the Ranger Station remained vulnerable and could be excessively depleted to a level beyond recovery.

Although TRNP has been subjected to poaching for several years already, the current density of trochus (23.64 ind 100 m^{-2} or 2 364 ind ha^{-1}) at the park is still much higher than the reported densities of trochus in seven South Pacific Countries between 1999 and 2007. Only the densities at French Polynesia had an average of 800 ind ha^{-1}, while the rest were between > 10 to < 500 ind ha^{-1} (Lasi 2010). At Site 4, where only empty shells have been found since 2008 (Jontilla et al. in press), its recovery from overfishing through recruitment derived from other parts of the park might be very slow because of the site's remoteness (Nash 1993; Crowe et al. 1997). The inability of trochus larvae to overcome barriers is manifested by its patchy distribution within its natural range, and its ability to thrive when introduced to suitable areas (Sims 1984; Gillett 1993; Nash 1993; Nash et al. 1995; Ponia et al. 1997; Zoutendyk 1997). While we do not disregard the potential of settling recruits being transported in from other parts of the park, the translocation of breeding adults (Gillett 1993; Bell et al. 2005) along with effective surveillance to arrest all forms of extractive activities could be the best option in restoring the extinguished population at Site 4. The current density of 3.5 – 40 ind ha^{-1} at complex habitat of Sites 5 and 6 (Rasa Island and Binduyan) is comparable to the overfished trochus (11 ind ha^{-1}) in Solomon Island, but were much lower than in other countries in the Pacific (Lasi 2010). Trochus density in an optimally fished area could range 100 – 300 ind ha^{-1} (Foale 1998), and this is much higher than in Sites 5 and 6.

In TRNPs seven permanent monitoring stations, the increasing trend in the mean sizes of trochus from 2006 to 2009 could be the result of poaching on the shallow area, harvesting most small sized-shells as reflected in 2007 data, and the possible increase in growth at reduced density. The significantly smaller average size (± sd) of trochus in this study (89.38 ± 10.52 mm) than in 2009

(93.50 ± 15.31 mm) appeared to be an effect of having many smaller individuals in the samples, a possible effect of increased in number of transect lines and area covered during the survey, effect of recruitment, or reduced exploitation in some sites. The sizes of trochus measured from confiscated shells in 2007 could form some bias in the data as the site of harvesting is not known. However, this data could be a representative of trochus from the shallow subtidal habitats, given the abrupt decline in abundance in 2008 which relatively remained low until 2010.

This study has shown how exploitation can severely reduce the trochus population at a shorter period and how can continued exploitation prevent the recovery of the trochus population in MPAs in spite of the presence of recruits. As this study only covers three of the many MPAs in Palawan, periodic assessment of trochus and other reef invertebrates in other MPAs within the country and understanding what factors affect the inability of trochus and other reef invertebrates to recover in long term managed reserves can help develop possible solutions and evaluate the effectiveness of such management action.

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Chapter 3

BREEDING AND SEA RANCHING OF *Trochus niloticus* IN PALAWAN, PHILIPPINES

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Alastair Grant and Jennifer A. Gill

ABSTRACT

Rebuilding the populations of the threatened reef gastropod *Trochus niloticus* by releasing hatchery produced juveniles is one of the many management options in conserving its depleted stock. We evaluate the trochus breeding and sea ranching project carried out by a private company in Palawan, Philippines from 1999 – 2008. Breeding and production of juveniles were successfully undertaken but the survival of the released juveniles was very low as a result of harvesting by local fishers and for a high incidence of natural mortalities. Sea ranching projects will only be successful if survival of the released juveniles can be increased. There is also a need improve the survival of broodstock which suffered from high mortalities during recovery from induction of spawning.

Keywords: breeding, Palawan, restocking, *Trochus niloticus*

INTRODUCTION

Many of the world's fishery resources are heavily exploited (Anonymous 1980, Anonymous 1987; Hodgson and Liebeler 2002), but with the rapidly increasing human population (UN 2004), fisheries production must be increased substantially (Delgado et al. 2003). As a consequence, there is a need to rebuild currently overexploited fisheries or improve the productivity of some healthy fisheries (Bell et al. 2006). Sea ranching is a stock enhancement system involving the release of hatchery produced juveniles into the wild followed by harvesting of the released animals at larger size when natural recruitment is low or absent as a result of overharvesting or lack of spawning or nursery habitat. It is most effective when the survival of the released juveniles is high (Bell et al. 2006 and references therein). If properly managed, this method of augmenting natural stocks could play an important role in coastal fisheries by providing sustained livelihood and source of protein for the coastal communities (Støttrup and Sparrevohn 2007). After decades of research, many countries have developed effective techniques in stock enhancement and economic viability have been demonstrated for some species (Ungson et al. 1993; Moksness and Støle 1997; Davenport et al. 1999; Wang et al. 2006; Halldórsson et al. in press). In spite of these successes, the contribution of stock enhancement to global fisheries production remains small (Lorenzen 2008).

Trochus niloticus, commonly called trochus, is a commercially important but over-harvested reef gastropod (Bell et al. 2005; Lincoln-Smith et al. 2006; Lasi 2010). It is the most important non-fish marine resource in terms of export earnings for some small islands in the South Pacific (Heslinga and Hillmann 1981; Legata 1997; Pakoa et al. 2010). Trochus are highly valued for their meat as food and for their shells used for the production of mother-of-pearl buttons. Their populations have severely declined (ICECON 1997; Isa et al. 1997b; Purcell 2004; Lasi 2010), prompting some countries to declare it as a protected species (Floren 2003; Hoang et al. 2008; Ramakrishna et al. 2010). To revive the depleted trochus fisheries and to sustainably harvest the shells, stock management involving multiple management strategies have been suggested (Nash 1988; Nash et al. 1995; Amos 1997; Lee and Lynch 1997; Trianni 2002; Purcell 2004; Bell et al. 2005; Dumas et al. 2010).

Mass production of trochus juveniles and restocking is one of the many conservation strategies for trochus. Successful seed production of trochus started in the 1980s (Heslinga 1980, Heslinga 1981a; Heslinga 1981b; Heslinga and Hillmann 1981) and from there towards the early 1990s, many countries in the Pacific became interested in establishing trochus hatcheries for sea ranching or stock enhancement (Nash 1989; Lee and Amos 1997). Enhancing of depleted trochus population through the release of mass produced juveniles had been tested (Crowe et al. 1997; Lee and Lynch 1997; Purcell et al. 2004; Purcell and Cheng 2010), however, hatchery produced juveniles suffered high mortalities upon released in the wild mainly because of predation (Isa et al. 1997a; Crowe et al. 2002), even when released as sub-adults (Villanueva et al. 2010). Trochus mass production is capital intensive so many hatchery projects were dependent on external funding (Lee and Amos 1997; Lee 2003; Lober et al. 2003) making trochus juvenile reseeding an uneconomic stock enhancement method, unless there is high survival both in tanks and in the sea, and if production costs can be reduced (Nash 1993). As early as 1992, conservation efforts for trochus in Palau were diverted back to conventional methods because these were more cost effective than mass production and reseeding technique (Amos 1997).

In Palawan, Philippines, reseeding of hatchery produced trochus juveniles was initiated by a Japanese corporation called Iris Marine Development Corporation (IMDC) in the late 1990s. The corporation started to breed trochus in September 1999 with the aim to (1) produce trochus juveniles for stock enhancement purposes, and (2) harvest the trochus released in the wild for the production of mother-of-pearl buttons. This sparked interests for some agencies in other parts of the country to also engage in trochus breeding and restocking in the early 2000 (Gapasin et al. 2002; Gallardo 2003; Pastor and Juinio-Meñez 2005). For example, the Aquaculture Department of the Southeast Asian Fisheries

Development Center (SEAFDEC/AQD) started trochus propagation in 2000 using some broodstock obtained from the IMDC hatchery in Palawan. As a result, the project produced thousands of diet tagged juveniles for release. In 2003, 3 000 diet tagged juveniles were brought to Palawan for release in Sabang Reef Sanctuary with the help of training course participants on Fish Sanctuary and Trochus Shell Resources Management sponsored by the Bureau of Fisheries and Aquatic Resources-Fisheries Resource Management Project (BFAR-FRMP) (Gallardo 2003). The SEAFDEC/AQD trochus breeding project however, was short-lived because of problems with funding and species prioritisation (Gapasin pers. comm.). The IMDC, after nine years of breeding and releasing of juveniles, ceased the operation and donated the hatchery complex to the Western Philippines University (WPU) which continued the breeding and restocking program since 2009 through the funds coming from the Department of Science and Technology-Philippine Council for Aquatic and Marine and Research and Development or DOST-PCAMRD (Gonzales et al. 2009; Avillanosa et al. 2010). In this study, we examine the accomplishments and problems related to breeding and sea ranching of trochus in Palawan, Philippines. Specifically, we described the (a) induced spawning techniques employed, (b) yearly eggs and larval production, (c) growth of juveniles in tanks and volume of yearly releases, and (d) status of the areas of release.

METHODS

The hatchery facilities and survey sites

The hatchery complex of IMDC sits in a 1.5 ha property (10^o 01.379' N and 119^o 6.337' E) along the coast of Binduyan, a northern village in Puerto Princesa City (PPC), Province of Palawan. The hatchery is equipped with a generator, two sand filter machines, 16 grow-out concrete tanks and 16 wooden tanks for the culture of natural food, and staff houses (**Figure 3.1**). The estimated value of the hatchery facilities in 1990s was PhP 150 million (or US$ 3.5 million at current rate of US$ 1 = PhP 43). The intertidal area in front of the IMDC hatchery (**Figure 3.1, middle right**) and in Sabang Reef (**Figure 3.1, bottom right**) were two of the three release sites for hatchery produced trochus.

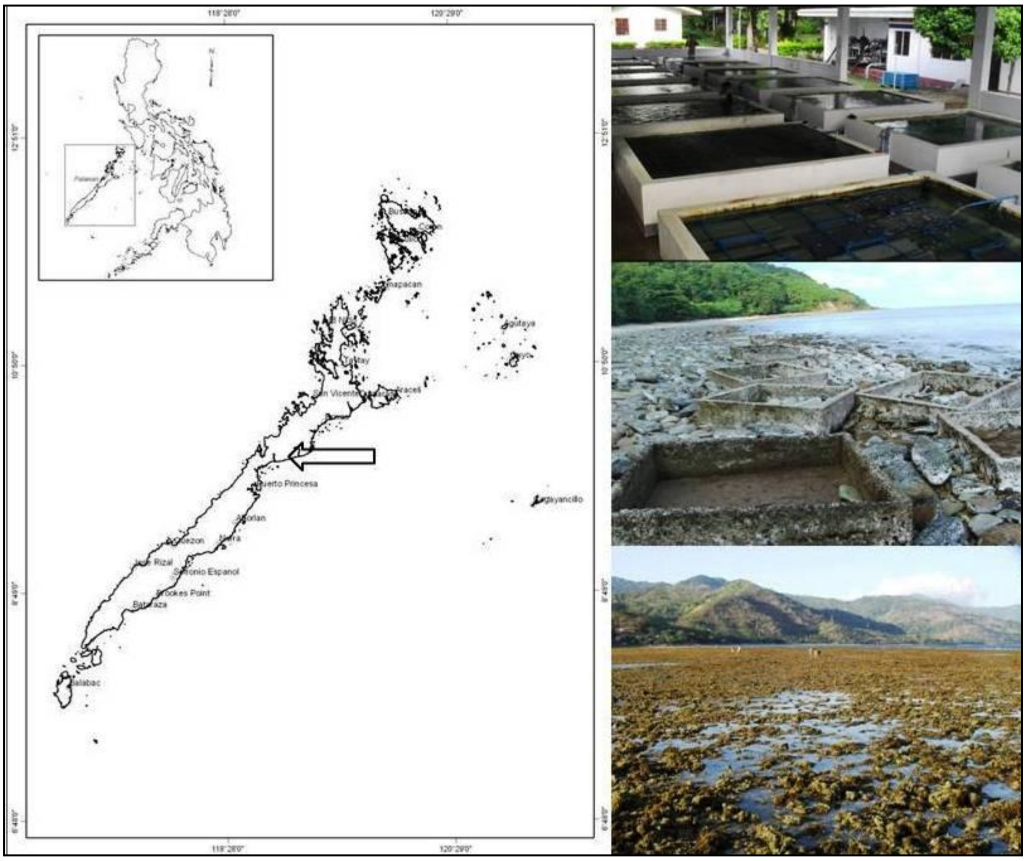

Figure 3.1. The study sites with the map of Palawan (left) indicating the location of hatchery (arrow) in the village of Binduyan, Puerto Princesa City (PPC). The hatchery facilities such as indoor tanks (upper right) and the unused intertidal tanks exposed at low tide (middle right). The continuously gleaned intertidal flat of Sabang Reef Marine Sanctuary during low tide (lower right). Sabang Reef is about 6 km southwest of the hatchery.

Induced spawning activities, larval production, growth of juveniles and volume of released shells

The induced spawning techniques employed by the IMDC was obtained or written through personal accounts of the second author who worked at IMDC as hatchery staff during the years 1999 – 2008. The Record Book of IMDC was examined to summarise the company's breeding accomplishments between 1999 and 2008. The extracted information for each induced spawning occasion include the number of breeding males and females; average size of the breeders per induced spawning; total number of eggs; fertilised eggs 12 h after hatching; and number of swimming larvae 36 h after spawning. Data on growth of juveniles in tanks were taken from a display of specimens at particular age.

Survey of the released sites

The sites for release of juveniles were the nearby village of Tinitian under the jurisdiction of the municipality of Roxas, about 15 km, northeast of IMDC hatchery. Another site is the intertidal reef flat of Sabang Reef, about 6 km southwest of IMDC hatchery. The intertidal area of Sabang Reef is part of the 40-ha village marine sanctuary of Binduyan (**Chapter 2**). Other releases were right on the reef flat adjacent the hatchery complex.

Visits and observations at two of the three release sites such as Sabang Reef ($10^{o}0.649'$ N and $119^{o}4.396'$ E) and the reef slope near the IMDC hatchery (10^{o} 01.379' N and 119^{o} 6.337' E) were conducted in May 2010 (**Figure 3.1**). The abundance of the trochus in Sabang Reef was assessed by transect surveys. Each transect measured 2 x 20 m. Trochus found within each transect were measured for their maximum basal diameter with sliding callipers to the nearest 0.1 mm and were immediately returned in their natural habitat. There were 40 and 16 transects at the reef flat and slope respectively. In a reef slope near the IMDC hatchery, no trochus were found after several snorkelling activities so we decided not to conduct the transect survey. Instead we asked occasional gleaners to hand in any trochus they found along the shore.

Data analyses

Descriptive analysis was used to present the yearly accomplishments of the IMDC. Total number of spawners used per induced spawning and the percentage of male and female that spawned were determined. The number of female spawners that spawned partially was also noted.

Exploratory analysis was conducted by plotting the percentages of breeding females and fertilisation rates against the percentage of breeding males to determine any sign of trends. The mean diameter of all breeders was also plotted with the average number of eggs per female produced per induced spawning. Yearly number of eggs produced, and fertilisation and hatching rates were determined. Fertilisation rate was obtained as the ratio between the number of fertilised eggs and the total number of eggs collected. Hatching rate was computed based on the number of larvae that hatched and the total number of fertilised eggs at each spawning event. Because of problems with recording, growth of juveniles in tanks was based on the specimens displayed on a glass cabinet showing the average size of trochus at particular age. The data on size at age of juveniles were plotted and a trend line was added to determine the regression equation. The total number of larvae and juveniles produced and released by IMDC was noted and graphically presented.

RESULTS

Induction of spawning

The collection of spawner and the induction of spawning were timed with the highest spring tide which coincide with the phase of the moon – a time when trochus are more visible on the reef as claimed by the fishermen. The whole process of induction from the arrival of wild spawners at IMDC hatchery until hatching of larvae took about 66 hours, with the bulk of activity on the second night when breeders are induced to spawn. Upon arrival (usually at 18:00 hours), the shells were cleaned and kept for 24 hours in aerated tank wrapped with opaque covering to create a dark environment inside the tank. Induction of spawning began on the second night at 18:00 hours by transferring the wild trochus breeders into a spawning tank with continuously flowing multi-stage filtered and UV-treated sea water for a group of wild trochus placed in a spawning tank. Generally male trochus spawned first at least an hour after the induced spawning, while the females spawned 1 – 2 hours after the first male had spawned. Some breeding males were transferred to a separate container with UV-treated seawater where they continue to release sperm. Trochus induced to spawn formed a cluster or some individuals crawled up the side of the tank until reaching the water level. Any breeding female was quickly transferred into a separate container filled with UV-treated seawater where it continued to release the eggs. Only one spawning female was placed per container, and the spent female was transferred to an aerated tank after spawning. The eggs in each container were then mixed with a 5 ml of seawater containing sperm. These activities lasted until 24:00 or 01:00 hours (or a period of about 6 hours). Twelve hours later, numbers of fertilised eggs were estimated and after 24 hours the numbers of hatched larvae were counted by volumetric method. If none of the trochus spawned, a shell (preferably male) was sacrificed and the sperm were mixed into the water in the spawning holding tank to stimulate other individuals to spawn.

Between 1999 and 2008, the IMDC successfully induced 60 groups with a total of 2 613 individuals to spawn. The numbers of breeders per induced spawning activity during the year 1999 – 2008 ranged between 18 –85 individuals with a mean (± sd) of 44 ± 16 ind. The average diameter and weight of breeders per group ranged between 49.0 – 113.3 mm, and 191.4 – 464.0 g. The general average (± sd) diameter and weight of all spawners were 93.0 ± 9.78 mm and 308.4 ± 66.6 g respectively.

The highest numbers of trochus subjected to spawn were between 2003 and 2006 comprising 74 % of all breeders. From 1999 – 2002 and 2007 – 2008 attempts to induce trochus to spawn occurred between 1 and 4 times a year, reflecting a very low number of spawners used. The highest percentage of breeding males (up to 18 %) and females (up to 7 %) in relation to the total number of breeders used per induced spawning event were between 2002 and 2007 (**Figure 3.2**). In total, 340 (13.0 % of 2 613 ind) male individuals were successfully induced to spawn and these stimulated a total of 110 (4.2 % of 2 613 ind) females to continuously release eggs. A small percentage (1.3 % of 2 613 ind; or 23 % of identified females) of females discontinued spawning when transferred from the induced spawning tank into a separate spawning container. Failure to induce trochus to spawn were not uncommon. There were 22 (37 %) and 25 (42 %) occasions that no male and female breeders responded out of 60 induced spawning events for a period of ten years.

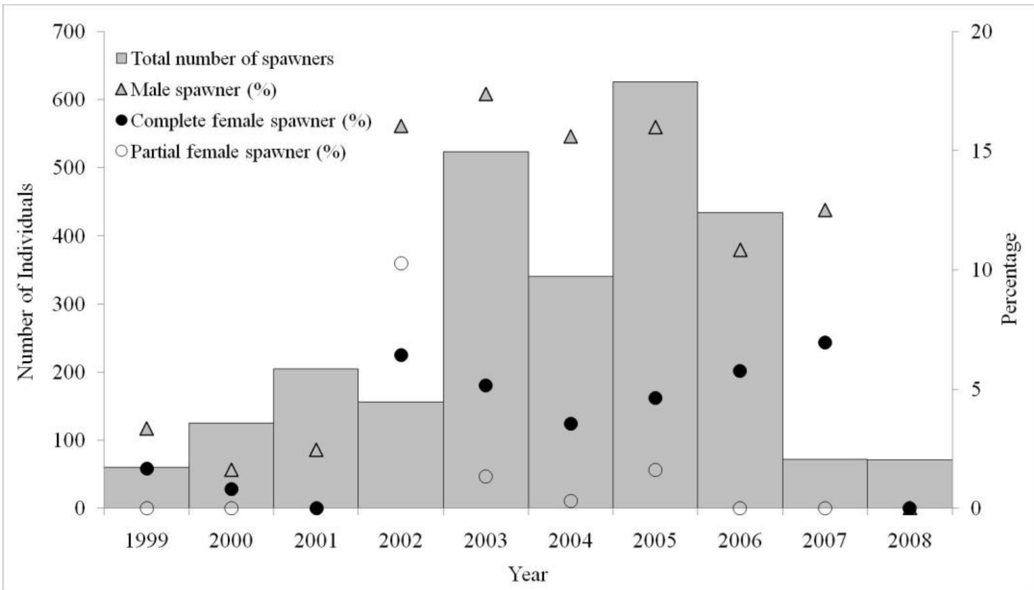

Figure 3.2. Yearly total number of wild trochus spawners induced to spawn during the year 1999 – 2008, and percentages of breeding males and females trochus that underwent complete and partial/discontinued spawning at IMDC hatchery in Binduyan, PPC.

It appears that the percentage of breeding females and fertilisation rates were not related to the percentage of breeding males. Percentages of breeding females remained between 0 – 20 % even with increasing percentage of breeding males. Also, fertilization rates remained constant between 30 – 90 % with an average of about 80 %, even with increasing percentage of breeding males (**Figure 3.3**).

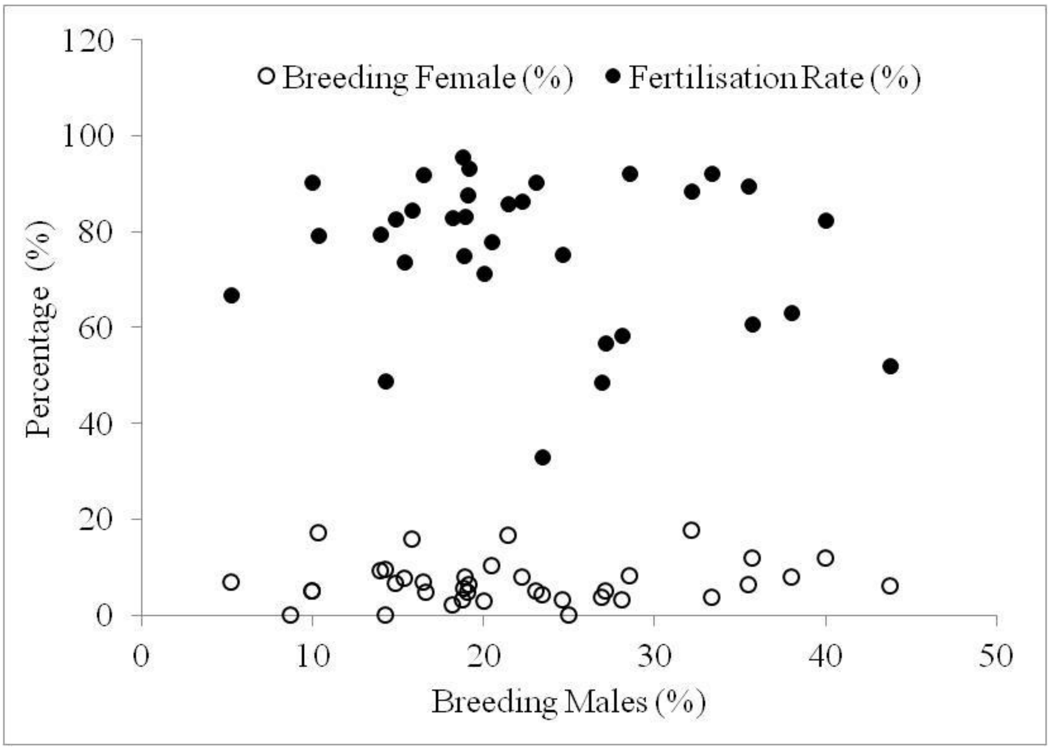

Figure 3.3. Percentages of breeding females and fertilisation rates in relation to percentages of breeding males during the induced spawning of wild trochus at IMDC hatchery in Binduyan, PPC between 1999 and 2008.

There was no record on the sizes of each female that spawned successfully. So the average number of eggs per female per spawning event was plotted against the average size of all breeders within that event. There is no relationship between the average size of all breeders and the average number of eggs (**Figure 3.4**).

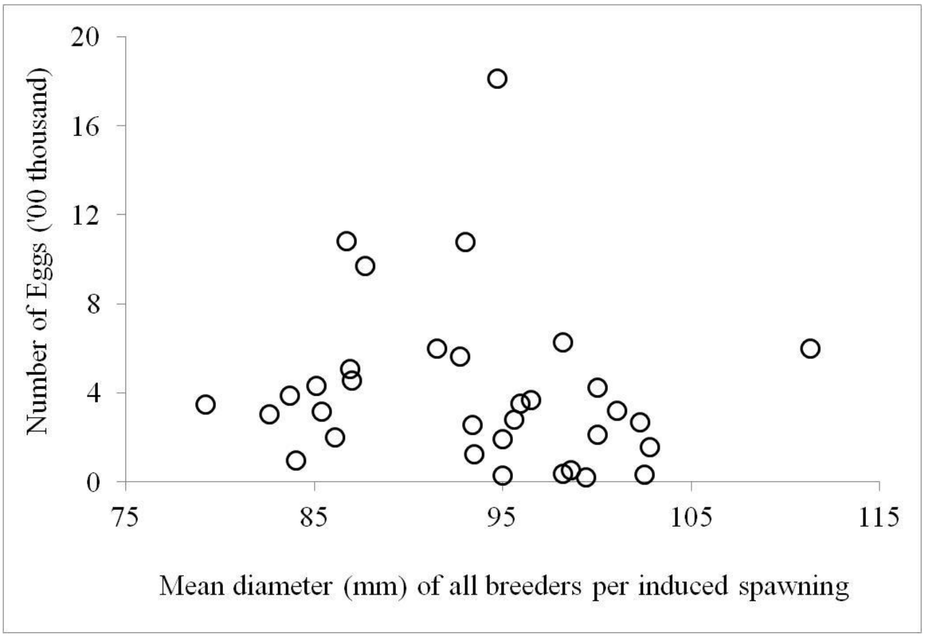

Figure 3.4. Relationship between the average numbers of eggs per female per spawning event against the mean diameter of all breeders induced to spawn by IMDC hatchery in Binduyan, PPC between 1999 and 2008.

Yearly egg and larval production

Between the years 1999 – 2007, the peak of egg production was during the years 2003 – 2006. In total, about 50 million eggs were produced. Although the numbers of fertilised eggs were monitored at the start of the project, only in 2003 did the management started to monitor the number of hatched larvae. There were only two induced breeding attempts in 2008 and were all unsuccessful (**Figure 3.5**). The peak in egg production coincided with the total number of spawners (**Figure 3.2**).

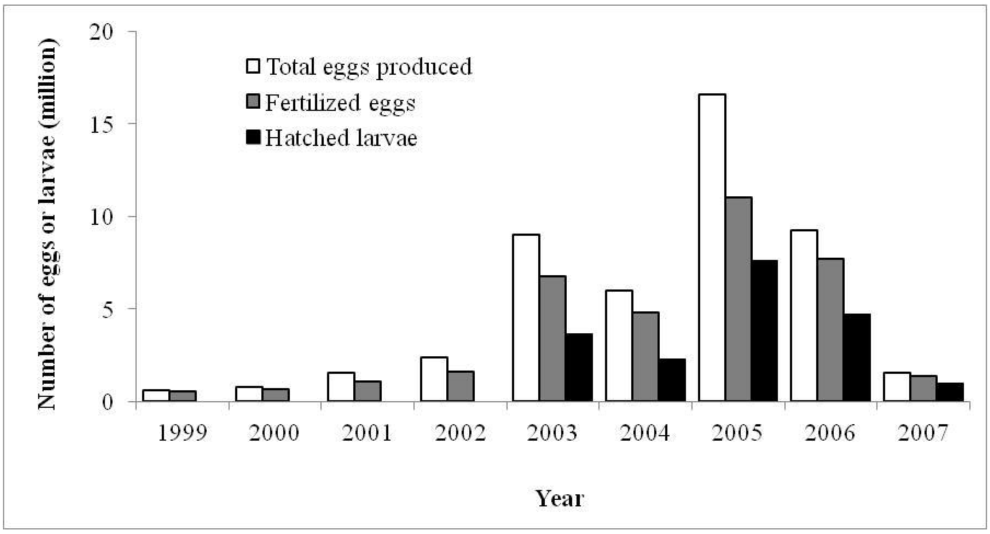

Figure 3.5. Yearly total number of eggs and hatched larvae produced by induced spawning at IMDC hatchery between 1997 and 2007. Induced spawning attempts in 2008 were unsuccessful.

The yearly average fertilisation rate ranged between 70 and 90 % with a general average of 80 %. Yearly average hatching rates were a little lower, ranging between 50 and 70 % with an overall average of about 60 % (**Figure 3.6**). This suggests that out of 50 million eggs collected, 24 million larvae were produced that were either raised in the hatchery or were directly released into the sea.

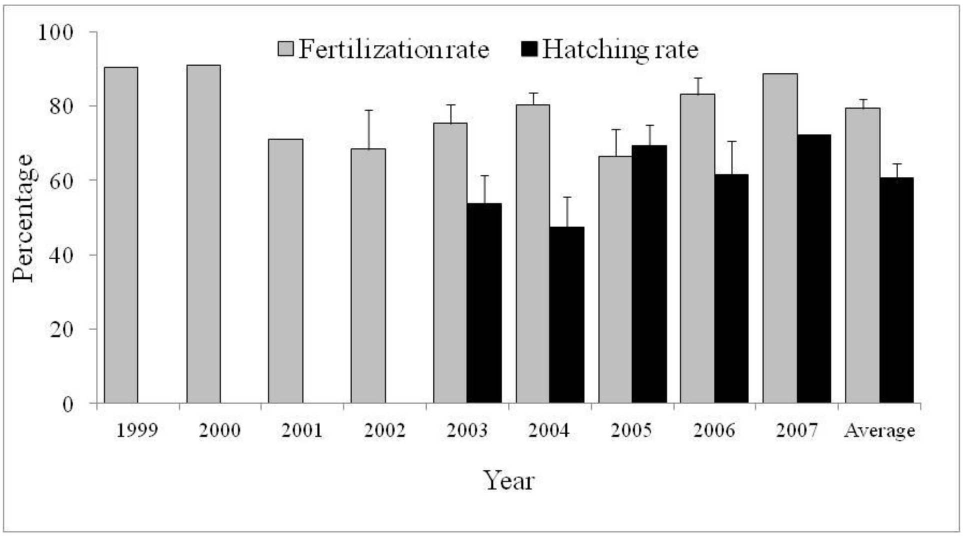

Figure 3.6. Yearly fertilisation and hatching rates of eggs produced by induced spawning of wild trochus at IMDC hatchery between 1999 and 2007. Error bars are standard error of the mean.

Growth of juveniles in tanks and volume of yearly releases

Trochus reared in indoor tanks were mainly feed on *Navicula* spp., a diatom that was grown on corrugated plates through the application of inorganic fertiliser. The duration of rearing in indoor tanks varied largely but we found it hard to trace the growth of one group of trochus because of problems with recording and the periodic transfer of trochus from one tank to another once the food was over grazed. Nevertheless, the average size of trochus at a particular month (**Figure 3.7**) displayed in a glass exhibit at the hatchery reflected an exceedingly slow growth reaching only around 55 mm in three years.

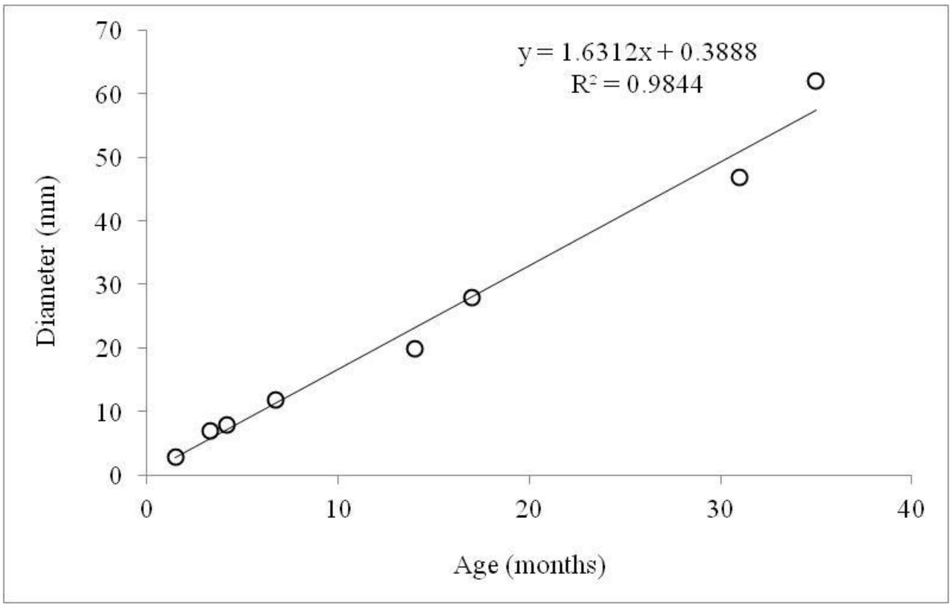

Figure 3.7. Approximate size at age of trochus reared from larval stage up to 36 months (3 years) as displayed in a glass exhibit at IMDC hatchery.

The IMDC had released trochus juveniles and larvae into the wild between 2002 and 2006, with the highest number of record for juveniles in 2005 and for larvae in 2003. In 2002, the number of released juveniles (25 – 29 mm) in the wild was about 5 000 individuals. Between 2002 and 2006, more than one million juveniles and sub-adults with size ranging between 3.3 – 35.7 mm were released, representing about 2 % of the 50 million total number of eggs produced. Of these nearly half a million were released at the intertidal area in front of IMDC, 0.4 million at Sabang Reef and 0.2 million at Tinitian Fish Sanctuaries. As to the larvae produced, about 6 million (or 12 % of 50 million eggs) were released at IMDC between the years 2002 – 2006 (**Figure 3.8**).

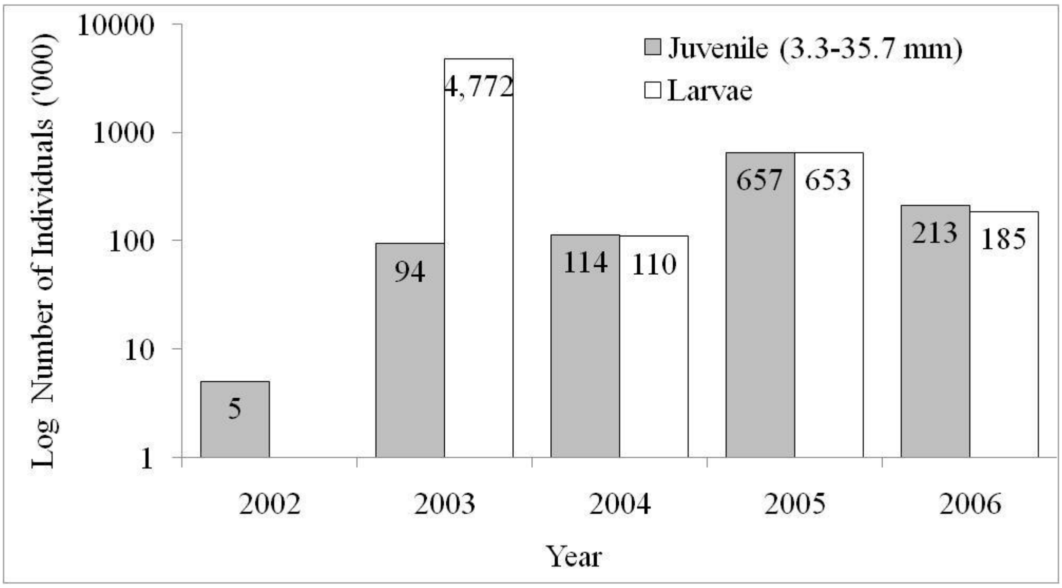

Figure 3.8. Numbers of trochus juveniles and larvae produced at IMDC hatchery between 2002 and 2006 that were released in Tinitian, Sabang Reef and in front of IMDC hatchery.

Status of the sites where hatchery-produced trochus were released

A visit to one of the sites of release (Sabang Reef – a village marine sanctuary) indicated the suitability of the intertidal habitat for both juveniles and adults, but there were gleaners on the exposed reef and only natural recruits were encountered. The wide inner rocky reef flat and the outer narrow strip of boulder were covered with patches of turf seaweeds that can serve as perfect refuge for trochus against desiccations and predators.

During one of our visits at Sabang Reef, about 20 - 25 gleaners (children and adults) were walking on the exposed reef searching for any edible marine life such as shells, echinoderms and octopus. Although an agreement was made between the company and the village people that trochus should not be harvested from the area of release, the staff during their random monitoring of gleaner's catches, found trochus juveniles. There were no fishers at the subtidal reef slope. In a reef area adjacent to the hatchery, only one or two gleaners were noted in a week.

The density of natural trochus recruits at the intertidal was 1.62 ind 100 m^{-2} (CI at 95 %: 0.81 – 2.44 ind 100 m^{-2}). At the reef slope, the density was only about 0.3 ind 100 m^{-2}. Six months prior to the survey, the WPU-Samong Farm had released about 2 000 juveniles (8 – 15 mm) at Sabang Reef but none of the tagged shells were encountered during our survey.

The size of trochus at Sabang Reef increases with depth. At the intertidal area, trochus had an average size of 42.94 mm (CI at 95 %: 39.14 – 46.73 mm, n = 17) while a single adult encountered at the subtidal reef measured 75 mm in diameter. In a reef slope adjacent to IMDC hatchery, the mean size of four sub-adult trochus handed in to us by the gleaners was 34.85 mm (CI at 95 %: 20.44 – 49.26 mm, n = 4). At the subtidal area, we did not find any large trochus during the visual survey.

DISCUSSION

There are a number of methods used to induce trochus to spawn (see Isa et al. 1997; Dwiono et al. 1997). The method employed by IMDC was quite laborious and could have caused some distress to spawning animals. The method could be improved by allowing the eggs to flow out of the tank and collected using an egg-catching device (Isa et al. 1997a), the same method used in collecting eggs of groupers in tanks (Dolorosa 1996). The induced spawning method employed by Dwiono et al. (1997) which was found successful when conducted 5 – 7 days before the new moon differed from the method of IMDC. The method of Dwiono et al. (1997) involved the cleaned shells being subjected to strong aeration in small plastic jars filled with UV-treated sea water for 6 – 8 h. In the evening, the shells were moved to a spawning tank filled with UV-treated water and left in darkness without aeration to allow the shells to spawn, unlike in IMDC where female trochus were grabbed quickly at the first sign of egg release. UV-treated seawater can deactivate microbes (Hijnen et al. 2006) thus promoting high hatching rates._The fertilised eggs were then collected with the use of nets rinsed with filtered sea water. This method can help reduce the disturbance to spawning females which can possibly result to an increase number of collected eggs. However, in this method, spawning individuals cannot be identified, and cannot be segregated in cases where induced spawning is repeated the following day. It could be advantageous to determine the sex and the weight of the breeders before the induced spawning. This can be done by examining under the microscope the water they extrude upon pressing their operculum for any presence of gametes (Dobson and Lee 1996). If induced spawning is repeated the next night, re-examining the extruded water of identified females for any presence of gametes, and changes in weight could help in identifying spent females. In that case, stress for spent females can be reduced.

The idea of subjecting trochus to free flowing sea water to induce spawning could have possibly come from the information that natural spawning synchronised with tide (see Nash 1993).Under captive condition, the spawning activities of females coincided with lunar cycle but not among male trochus (Nash 1985).

The potential fecundity of trochus is proportional with size (Heslinga 1981b; Bour 1989) but the number of eggs released during the spawning in hatchery (realised fecundity) does not increase with the size of the breeders (Nash

1985; Kitalong and Orak 1997). There has been a conflicting idea about the spawning behaviour of trochus. According to Hahn (1993), trochus are complete spawners because of a single cohort of gametes developing at a time, however, Pradina et al. (1997) suggest that trochus are capable of complete or partial spawning and stress can cause lower hatching rates (Dwiono et al. 1997). The lack of relationship between the number of eggs and size of spawner may indicate that incomplete spawning is taking place during induced spawning. It was observed during the induced spawning that breeders kept overnight by fishermen were less likely to spawn than those which were immediately delivered in the afternoon on the day of collection. Keeping shells overnight in unsuitable conditions can cause stress on the animals which could result to a failure in spawning (Nash 1989). If this is the case, it is important that the contracted fishermen are given guidance in handling trochus breeders from collection and keeping on the boat while transporting to the hatchery. A simple method in handling trochus to minimise stress is outlined by Lee and Ostle (1997).

The absence of the relationship between the percentage of breeding males with females could be due to the following reasons: 1) Each mature female can spawn at least 3 times in two years and only about 12.5 % of the female in the population may spawn each month (Hahn 1993); 2) Because of differential spawning behaviour in the wild (Nash 1993); and 3) The high number of spawning males could be typical in the wild to ensure high fertilisation rate, considering that fertilisation rate for other invertebrates decline with increasing distance between breeders (Babcock and Keesing 1999; Lundquist and Botsford 2004). Sex identification by examining the gametes excreted by the shells upon pressing the operculum (Dobson and Lee 1996) might help answer these questions, although this method of sex identification may only apply for mature individuals that are ready to spawn.

After the first induced spawning, the breeders were released in the reef slope near the hatchery but the shells tended to move outside the restricted area allocated by the village to IMDC. Within a month from the date of release, some breeders were recovered (by snorkelling) more than 100 m from the point of release. They moved somewhat parallel the shore line but gradually moving towards the deep. Because of fears that the released breeders would be harvested by the locals, the broodstock used in the succeeding induced spawning events were kept in indoor tanks. Trochus in the wild feed on a variety of seaweeds (Burhanuddin 1997; Isa et al. 1997a; Lambrinidis et al. 1997b) which were not available when rearing the broodstock in indoor tanks. As a consequence, gradual mortalities occurred in tanks possibly because of the compounded effect of starvation and water fouling caused by rotting meat of other dead shells. The experiments of Li et al. on the bacterial resistance (2009a) and food deprivation (2009b) on Pacific Oyster *Crassostrea gigas* before and after spawning revealed that post spawning oysters have significantly reduced hemolymph antimicrobial activity leading to either higher mortalities or prolonged post-spawning recovery. This led them to conclude that the energy cost of spawning compromises the immune and metabolic responses of oysters, leaving them more vulnerable to pathogens.

The practice of keeping the shells in indoor tanks after induced spawning reduces their survival, so is potentially detrimental to the natural trochus breeding population. The number of hatchery produced juveniles may not compensate for the number of natural recruits that would have been produced by those adults if they were in the wild. This reduces the number of potential breeders in the wild and the potential level of natural recruitment. It is suggested that after induced spawning, trochus breeders must be released in a well managed sanctuary to prevent harvesting, reduce natural mortalities and maximise their reproductive potential. Another option is to utilise the intertidal concrete cages to maintain breeders after the induced spawning. Although the use of intertidal tanks can potentially reduce mortality rates because of constant water exchange, natural food growing inside the tanks could be limited. Finding out if adult trochus can survive and naturally reproduce in intertidal concrete tanks could be an interesting topic to investigate.

Generally, trochus become sexually mature upon reaching a size of > 50 mm at two years old (Nash 1993). However, with adequate feeding and good management in indoor tanks, trochus can grow as fast as 5 mm in a month (Heslinga 1981b; Nash 1985; Hoang et al. 2008), reaching a sexually mature size (> 50 mm) in less than a year (Heslinga 1981b). By contrast, trochus growth at IMDC hatchery was much slower, reaching only a size of > 50 mm in 3 years time. Individuals did spawn successfully after attaining such size in 3 years suggesting that sexual maturity is size dependent. While 44 mm female trochus believed to be stunted were reported to spawn (see Nash 1993).

The exceedingly slow growth of trochus in indoor tanks could be due to limited nutrition, fouling of water or rearing of mixed sizes. Trochus juveniles of 20 000 individuals with a size range of 4 – 7 mm can graze out an area of 6.5 m^2 diatoms in a week (Dwiono et al. 1997) which makes it difficult to supply given the limited number of algal growing tanks. Dwiono et al. (1997) further noted that when there is shortage of food juveniles began to crawl up the side of the tank reaching beyond water level and they die because they were incapable of returning to the water. Under laboratory condition, low dissolved oxygen and high levels of ammonia have been found to cause a reduction in growth of abalone (Harris et al. 1998; Harris et al. 1999), while low dissolved oxygen affects the growth and survival of scallops (Chen et al. 2007). The mass mortalities of trochus kept in aquaria with daily exchange of water was attributed to the absence of aeration (Rao 1937). Trochus juveniles can withstand more than five months of limited food in cages set in indoor tanks (**Chapter 5**), so it is unlikely that limited food and undernourishment were the main causes of mortality for juveniles in indoor tanks. In the wild, feeding competition within species, leads to a tendency for larger individuals to dominate over the smaller individuals, as for the case of hermit crab *Pagurus bernhardus* (Ramsay et al. 1997). Rearing of mixed size juveniles in indoor tanks for a longer time may promote starvation and slow growth among the smaller individuals. However, for some species, increase in abundance of small individuals can significantly reduce the growth and survival of adult populations (see Underwood 1976; Marshall and Keough 1994).

Observations on the shell morphology of a 3-year old captive trochus showed the presence of flanged aperture (**Figure 3.9**) which is normally observed for a very large shell in the wild. Thick shells can be a characteristic of trochus occurring at higher densities (Nash 1985) because of slow growth due to limited food. A flanged lip to the shell indicates sexual maturity and cessation of growth in some gastropods as in the case of the humped conch *Strombus gibberulus* (Giovas et al. 2010).

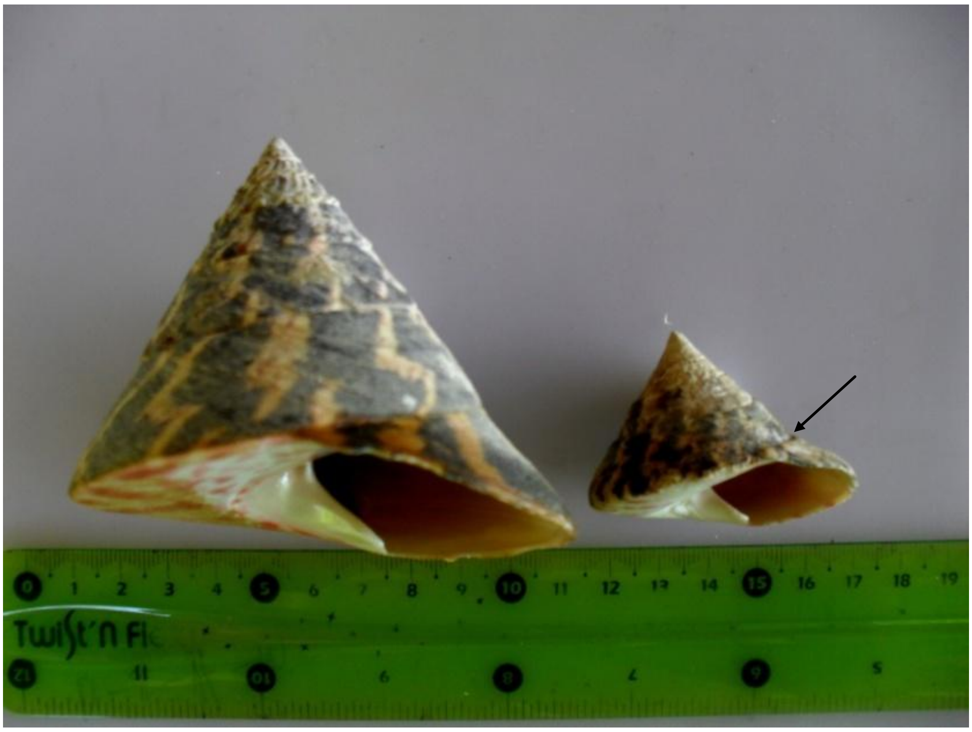

Figure 3.9. Large shell from the wild with no sign of lip projection (left); and a 3-year old hatchery grown trochus exhibiting a prominent flanged aperture (right) pointed by an arrow.

The diet of trochus may change as they grow bigger. Trochus in the wild feed on a large variety of plants and animals (Rao 1937; Burhanuddin 1997; Isa et al. 1997a; Lambrinidis et al. 1997b, Lambrinidis et al. 1997a; Soekendarsi et al. 1998; Ramakrishna et al. 2010) which is no longer met by the use of naturally grown algae in the laboratory. This problem could be addressed by 1) gradual thinning of stock – this could be done by sorting out 5 mm juveniles for intermediate rearing in sea cages until a sub-adult size is reached; 2) setting up of extra tanks only for growing of natural food to satisfy the demand for food of the smaller individuals left after the thinning out process; and 3) acclimation of trochus to predators to increase survivorship after the release. Purcell (2001) have successfully increased growth of trochus by adding more food, while Hoang et al. (2008) used a variety of foods to enhance the growth of trochus juveniles. Also, a gradual reduction in density (10 – 22 mm at 100 ind m^{-2}; 25 – 40 mm at < 50 ind m^{-2} and 40 – 50 mm at < 10 ind m^{-2}) as the shell size increases have been recommended by Hoang et al. (2007). Intermediate culture in cages until sub adult size was attained had been tested but requires further improvements (Nash 1989; Purcell 2001; Amos and Purcell 2003).

As reproduction only occur upon reaching the size of > 50 mm irrespective of age (Heslinga 1981b; Nash 1993), it is important to consider the type of habitat for release in order to shorten the time between releasing of juveniles and the time at which they may start to reproduce. Trochus released in habitats that can favour fast growth are more likely to survive and reproduce earlier than those released in unfavourable habitats. Colquhoun (2001) and Castell (1997) had found that intertidal habitats with boulders have higher densities of trochus than the flat rocky substrate. The release sites of IMDC were highly suitable for trochus juveniles, given the presence of fast growing recruits (5 mm mo^{-1}) on boulders and flat rocks covered with turf seaweeds (**see Chapter 6**).

Early observation on juveniles released in intertidal tanks showed faster growth than in indoor tanks but these were not been well documented in the log book, or if it was, we find it hard to connect the same groups and sampling dates. In a similar case, even if the recording was good, it would still be difficult to monitor the growth because the shells were unmarked and many had moved out of the uncovered tank after release. Trochus juveniles released in the wild can disperse up to about 5 m from the point of release after 45 days (Isa et al. 1997a).

Hatchery produced juveniles are more vulnerable to predators than their wild counterparts, but survival can be increased when acclimated to predators or natural substrate before restocking. Crowe et al. (2002) had released a considerable numbers of hatchery produced juveniles (size ranges: 6 – 12 and 16 – 22 mm) but suffered high mortality rates in just a month. Isa et al. (1997a) found that juveniles exposed to natural sea condition for 8 days can recover from an upside down position within 5 minutes compared to the newly released juveniles that would take about 30 minutes to resume an upright position. Cultured and wild trochus juveniles when exposed to a predatory gastropod *Thais tuberosa* showed a similar response by releasing white substances, but the cultured juveniles were slightly more active than the wild trochus juveniles (Castell and Sweatman 1997). This suggests that releasing the juveniles directly into the open intertidal area without proper acclimation can subject the juveniles under high predation pressure because of their indifference behaviour and inability to quickly adjust to the natural sea condition. Hatchery produced gastropods such as abalone *Haliotis rufescens* and queen conch *Strombus gigas* are also unaware of predators in the wild and could suffer high mortalities than wild juveniles (Schiel and Welden 1987; Stoner and Davis 1994; Stoner and Glazer 1998). Conditioning to fish predators has increased the survivorship of hatchery-produced crabs in the wild (Davis et al. 2005), while shelter acclimations have been found to increase the survivorship of hatchery-produced fish (Kawabata et al. 2011).

Without acclimation, the survival of hatchery produced trochus is very low. In her experiment, Castell (1993) estimated that at least 0.13 % hatchery produced juveniles of 4 – 15 mm in size would survive after six months in the wild. If we use the estimated value of Castell (1993) to estimate the survival of about 1 million hatchery produced trochus at IMDC hatchery upon released into the wild, only a thousand could have survived in 6 months, or 1 – 2 individuals could have survived after a year. A more discouraging 100 % mortality rates for 8 - 16 mm juveniles after 4.33 months was reported in Japan (Isa et al. 1997a). IMDC had released juveniles between 3.3 – 35.7 mm in diameter, so we assumed that some larger individuals could have survived from predation but not from unhampered gleaning activities at the sites of release. Anecdotal accounts of some gleaners include comments on the difficulty of removing tags from juvenile trochus before cooking. While random inspection done by IMDC staff on gleaned shells by the fishermen revealed the presence of trochus. The absence of large trochus even in a reef flat in front of the hatchery which also served as released site could be due to high predation on the released juveniles because gleaning and fishing were restricted on the area.

Natural recruits recorded during the survey were higher than the previous studies but much lower than the reported density in other sites. Survey at Sabang Reef revealed the presence of natural recruits of trochus at 1.62 ind 100 m^{-2} which was a little higher than in 2002 (0.4 ind 100 m^{-2}) and 2004 (0.7 ind 100 m^{-2}) as reported by Galon et al. (2007). This difference could reflect a variation in intensity of survival of recruits in space and time rather than an effect of restocking given the daily presence of gleaners. At the same time, trampling on intertidal habitat can cause disturbance and can reduce abundance of many species (Brosnan and Crumrine 1994; Brown and Taylor 1999; Schiel and Taylor 1999). The recorded densities at Sabang Reef were much lower than 12 – 18 ind 100 m^{-2} in Orpheus Island, Queensland, Australia, where juveniles often occurs in a group of 2 – 4 ind m^{-2} (Castell 1997). The density of trochus juveniles (30 – 50 mm) at King Sound, Australia however varied between < 1 to > 5 ind 100 m^{-2} depending on the type of habitat, with larger juveniles (40 – 50 mm) commonly occurring at rocky habitats (Colquhoun 2001).

The claim that an increase in trochus density in the deeper (4 – 21 m) parts of the reef from 1.5 to 1.9 ind 100 m^{-2} during the years 2004 and 2007 was an effect of stock enhancement (Becira et al. 2004; Galon et al. 2007) is difficult to prove because the authors have no evidence to support that the released juveniles have survived on the reef. The increase in density over time could be attributed to the limited depth at which the fishers can reach by breath-hold dive following the banning of the use of compressor fishing in the City of Puerto

Princesa as per City Ordinance No. 267. The presence of wild recruits and large trochus in the adjacent open-accessed unseeded Bitauran Reef (**see Chapters 2 and 6**) suggests that the recruits of large trochus recorded at the seeded Sabang Reef comprised the naturally occurring wild individuals and none of the reseeded hatchery produced juveniles have survived after the release.

It is still important to note that some conventional management methods are more effective in conserving trochus than the restocking of hatchery produced juveniles (Amos 1997). Reviving the trochus population in an area like the Sabang Reef may not require the seeding of hatchery produced juveniles but rather an effective fishing moratorium along with a combination of other traditional control measures such as catch quota and size limits (see Nash 1993). The active involvement of the local bodies is paramount in any conservation success (ICECON 1997; Pollnac et al. 2001; Purcell et al. 2004; Dumas et al. 2010; Pollnac et al. 2010), a good example of which is the nearby marine sanctuary of Caramay, a fishing village northeast of Binduyan, effectively managed by a cooperative (GEF 2011). Adapting this management scheme in a network of village managed reserves could have a wider impact in coral reef biodiversity restoration. The importance of habitat selection and the safety of the seeded stock against illegal harvest have been pointed out by Gomez and Mingoa-Licuanan (2006) as important factors in long-term giant clam restocking project in the Philippines. While Nash (1993) stressed out the potential of reseeding as a means of reviving depleted stocks on conditions that these released juveniles have high survival rates and become sexually mature in the wild, and these individuals are not harvested, but rather allowed to become part of the breeding population. Purcell and Lee (2001) although they agreed that broodstock translocation method is cheaper and simpler than restocking of juveniles, they suggested the latter over the former as a conservation method in reef areas that are unsuitable for trochus larval settlement but are favourable for juvenile growth and survival.

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Chapter 4

GROWTH, SURVIVAL AND POTENTIAL REPRODUCTIVE OUTPUT OF *Trochus niloticus* IN INTERTIDAL TANKS: IMPLICATIONS FOR MANAGEMENT

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ABSTRACT

The primary breeding practice for the threatened reef gastropod *Trochus niloticus* in the Philippines involves keeping wild-caught breeders in indoor tanks after induced spawning in captivity. However, mortality rate of wild-caught breeders is high in indoor tanks, reducing a chance to induce spawning repeatedly and terminating their potential contribution to natural recruitment when the shells are not returned to the wild. In this study we explored the survival and growth rates of wild-caught trochus at a density of 4 ind m^{-2} maintained in tanks at the intertidal zone rather than indoors, and determined their reproductive output by inducing them to spawn after 11 months in captivity. Trochus had 51.9 – 80.2 % survival rates in 11 months. There was a consistent increase on the average monthly diameter although the growth was very slow at an average increment of 2.2 mm in 11 months. The average monthly weights were characterised by drops and recoveries between months. Percentages of males that responded to induced spawning were five times higher than females. Females with an average (± sd) size of 87.2 (± 5.7) mm released an average of about 200 000 eggs. Average (± sd) fertilisation and hatching rates were 87 ± 6.3 % and 81 ± 4.5 % respectively. This study shows that the rearing of trochus breeders in intertidal tanks is feasible, and can be a cost effective means of enhancing a depleted population when combined with effective protection of marine sanctuaries.

Keywords: intertidal tanks, natural spawning, Philippines, *Trochus niloticus*

INTRODUCTION

Trochus niloticus, or trochus as it is commonly called, is considered the most commercially important reef gastropod in the tropical Pacific, where it forms the basis of a multi-million dollar mother-of-pearl industry (Heslinga 1981a; Heslinga and Hillmann 1981; Nash 1985; Hahn 2000). Historically, the shells occurred abundantly in shallow seaward reefs (Heslinga et al. 1984; Nash 1985) but very intense harvesting has resulted in serious population declines in many parts of the range (Nash 1985; Nash1993; Kelso 1996; Foale and Day 1997). In some countries, trochus exploitation has already been prohibited (Dwiono et al. 1997; Hoang et al. 2007) and a range of conservation measures have been undertaken for the species (Nash 1993; Foale and Day 1997; Isa et al. 1997a; Zoutendyk 1997; Foale 1998). In the 1980s, the International Union for the Conservation of Nature listed trochus as a commercially threatened species (Heslinga et al. 1984).

The Philippines was one of the world's major trochus producers (ICECON 1997; Hahn 2000) but, as with many trochus-producing countries, its volume of trade has declined (Hahn 2000; Floren 2003) due to unregulated harvesting (Gapasin et al. 2002; Gallardo 2003; Dolorosa et al. 2010; Jontilla et al. in press). As a consequence, its exploitation has been prohibited (DA 2001) and hatchery propagation and reseeding programs undertaken at the Southeast Asian Fisheries Development Center – Aquaculture Department (SEAFDEC-AQD) (Gapasin et al. 2002; Gallardo 2003) have been short-lived because of problems with funding and species prioritisation (Gapasin pers. comm.). At present only the Western Philippines University (WPU) in the province of Palawan is engaged in trochus propagation for restocking (Gonzales et al. 2009; Avillanosa et al. 2010), and this is only possible because of funding from the Department of Science and Technology-Philippine Council for Aquatic and Marine Research and Development (DOST-PCAMRD). Captive breeding is expensive and growing of trochus to a commercial size in indoor tanks is not feasible because large tank surface areas are required to grow food for the trochus (Nash 1993).

The development of techniques for mass production of trochus in captivity in the 1980s (Heslinga 1981a; Heslinga and Hillmann 1981) drew substantial interests in many other countries to attempt to artificially produce trochus and use these in restocking depleted populations (Dwiono et al. 1997; Isa et al. 1997b; Lee 1997; Lee and Amos 1997). Wild-caught trochus breeders held captive in indoor tanks with adequate water exchange and feeding spawned naturally and have a survival rate of 97 % mo^{-1} for 9 months (Heslinga and Hillmann 1981), however, for large scale production of trochus, induced spawning – a technique which uses varied artificial stimuli to initiate mass spawning is used (Dobson 1997; Isa et al. 1997b; Gapasin et al. 2002). There is an extensive literature dealing with the growth and survival of trochus juveniles in indoor facilities, sea cages and in the wild (Crowe et al. 1997; Gimin and Lee 1997a, b; Crowe et al. 2002; Amos and Purcell 2003; Purcell et al. 2004; Hoang et al. 2007; Hoang et al. 2008; Gonzales et al. 2009), but little is known on the fate of the wild breeders that have been collected and allowed to either naturally or artificially spawn in indoor tanks.

Trochus broodstock are mostly returned to the wild to recover after spawning (Nash 1989), but this is not the case in Palawan, Philippines. A Japanese company, Iris Marine Development Corporation (IMDC) that engaged in trochus propagation by inducing wild-caught trochus to spawn to produce and release hatchery-reared juveniles in Palawan between 1999 and 2008 had firstly released the wild-caught breeders in a reef slope (after the first induced spawning) in front of the hatchery that is restricted from fishing. However, during the periodic snorkelling to monitor the released shells, some individuals were recovered > 100 m from the point of release, reaching outside their designated boundaries from fishing. In such a case the breeders were recaptured and reared in indoor tanks to protect them from being harvested by fishermen. To avoid the danger of the released shells from being harvested, all other newly wild-caught breeders used in the successive spawning activities were kept in indoor tanks after the induced spawning, but these breeders did not survive for a long time under captive conditions (Cayaon pers. comm., **Chapter 3**). In nearly 10 years of operation, the IMDC reared in indoor tanks 2 348 newly wild-caught breeders after the induction of spawning. The same practice was adopted by the WPU-Samong Farm (WPU-SF) when IMDC closed down and donated their facilities to the university in 2008 (**Chapter 3**). The WPU-SF induced to spawn a total of 171 trochus in the summer of 2009 (Gonzales et al. 2009), after which the breeders were kept in indoor tanks and only 11 (6 %) individuals had survived after 4 – 5 months. Trochus breeders raised in indoor tanks require adequate food supply, good water quality and aeration which when not fully satisfied can affect their breeding condition, growth and life span (Nash 1989). Similarly, the success in the captive breeding of false clown fish *Amphiprion ocellaris* (Kumar and Balasubramanian 2009) and tiger prawn *Penaeus monodon* (Quinitio et al. 1996) broodstock was only made possible through suitable husbandry and conducive environmental parameters.

Rearing of trochus in intertidal tanks could possibly increase survivorship than when they are left for a long time in indoor tanks. The intertidal tanks are exposed periodically at low tide and the area is subjected to strong wave action, a habitat most preferred by trochus (Nash 1993). In intertidal tanks, trochus are in semi natural environment and will not suffer from low level of dissolved oxygen which often occurs in indoor tanks in times of electric power shortage or power interruption due to bad weather. Food supply is limited at indoor tanks, while trochus at intertidal tanks can graze on algae naturally growing on rocks or on suspended organic particles that settled inside the tank. Trochus juveniles kept in intertidal tanks grew faster than juveniles raised in indoor tanks (**see Chapter 3**).

This study explored the potential of keeping the trochus in intertidal tanks. Specifically, we examine the (a) growth and survival of broodstock for a period of 11 months, and (b) determine the fecundity of broodstock, and fertilisation and hatching rates of eggs produced by inducing trochus to spawn after 11 months in captivity.

METHODS

Study site

The study was carried out in September 2009 – August 2010 in intertidal concrete tanks at WPU-SF (10^o 01.379' N and 119^o 6.337' E) located in Binduyan, Puerto Princesa City, Philippines. The tanks were previously used for holding small-sized trochus juveniles (from indoor tanks) until ready for release in the wild but this has been stopped because juveniles tend to move out of the tanks and are difficult to recover (**see Chapter 3**). The tanks were exposed during extreme low tides and were subjected to strong waves during most months of the year except during the summer season (March to May). The shore is characterised by about 10-m wide strip of rocks exposed at low tide, while the subtidal area features a gradually sloping reef extending about 100 m from the shore before the drop off into the deep water. This type of habitat is considered favourable for trochus because they occur abundantly in seaward reefs with strong wave action (Nash 1985; Smith 1987; Nash 1993). Local residents confirmed the presence of large trochus in the past, but at present, only juveniles are occasionally encountered by gleaners along the rocky shore line.

Sources of trochus breeders

A total of 117 individuals were used in this study composed of 106 newly captured shells obtained by octopus fishermen from a nearby village, and the 11 remaining old breeders that had been kept in indoor tanks of WPU-SF for around 4-5 months after the induced spawning. Shell diameter for these 117 individuals ranged between 60.0 – 109.4 mm with an average (± sd) of 87.3 ± 9.7 mm.

Tagging of trochus breeders

Each shell was tagged following the procedure described in Nash (1985) and Lemouellic and Chauvet (2008) with some modifications:

1. Scrape off with a chisel any marine growth including a thin layer of periostracum on a 5 x 5 cm portion of the shell.
2. Dry the shell with cheese cloth; allow further drying for about 15 – 30 minutes while keeping them in an upright position in plastic trays lined with a damp cloth. Do not expose the shells to sunlight.
3. Apply a small amount of marine epoxy on the cleaned and dried shell's surface; put on the 3 x 1.2 cm pre-numbered commercial TZ extra strong adhesive yellow tape printed with a desired number using Brother P-Touch PT1260 printer.
4. Apply an additional marine epoxy to cover the sides of the tag leaving only the marked surface.
5. Keep the shells in plastic trays lined with a moist cloth for about an hour or until the epoxy has hardened. Do not expose the shells to direct sunlight.
6. With the use of a pencil, inscribe the same tag number on the nacreous layer inside the shell.

Intertidal tank preparation and management

Five intertidal concrete tanks with a dimension of 2.8 x 2.2 x 0.6 m were prepared by patching any large cracked flooring with nets and rocks to prevent the escape of the shells. Each tank received rock substrate covering about 60 – 70 % of the floor area, to provide refuge while still allowing the shells to be mobile within the tank. The shells were randomly distributed in five intertidal tanks at 4 ind m^{-2}. Coconut fronds as an additional substrate were added monthly starting from the second month of rearing (**Figure 4.1A**). Each tank was covered with painted steel matting overlaid with a plastic net (2 cm mesh size). The cover was fixed rigidly on each tank to withstand strong waves' impacts in times of bad weather. To protect the shells from extreme heat during the summer day low tides, each tank received a shade made of coconut fronds. The shade was removed at sundown or when the tide comes in (**Figure 4.1B**).

Sampling and monitoring of trochus breeders in intertidal tanks

The growth and survival of all individuals was recorded monthly. Collection of shells from intertidal tanks and measurement for size and weight usually took place in the morning and the shells were returned in the afternoon of the same day. Any loose tags were fixed, damaged tags were doubled with small tags, and pencil markings were re-inscribed every sampling event. Maximum basal diameter was measured with sliding callipers to the nearest 0.1 mm. Shell weight was recorded with digital scales to the nearest gram. The numbers of live and dead shells were noted every sampling period. Unrecovered trochus were presumed to have escaped from the tanks and were still alive unless an empty shell is recovered during the monthly 2 – 3 h snorkelling within the tank vicinities and in the adjacent reef slope. Any live shells recaptured outside the tanks were released back in the wild because the conditions (e.g. food abundance) inside and outside the tanks vary and the time of escape is not known.

Induced spawning of trochus

All individuals (n = 55) recovered on the 11th month of captive rearing were induced to spawn (**Figure 4.1 C, D, E, F**). In the induction of spawning, we followed the procedure of IMDC which have some similarities with the induced spawning methods mentioned in the studies of Dobson (1997), Dwiono et al.(1997), Isa et al. (1997b), and Gapasin et al. (2002). This method involved the cleaning of the shells and measurement of diameter before keeping them for 24 h in aerated 100 L capacity polycarbonate circular water tank wrapped with an opaque sack to create a continuous dark condition. After 24 hours, the shells were transferred to a 300 L capacity rectangular polyethylene spawning tank and continuously flowing two-stage filtered (Fuji pebbles/sand filter machine and one stage 50 or 100 micron polypropylene cartridge) UV-treated (Rei-Sea UVF-10) sea water was applied to induce spawning. UV-treated water deactivates microbes (Hijnen et al. 2006) which can reduce hatching rates of fertilised eggs.

Figure 4.1. The intertidal tanks (A & B); arranging of breeders in spawning tank (C); continuous observation for any sign of breeding females (D); trochus clumping with each other, the one at the centre with extended siphon (encircled) was a breeding male (E); counting of eggs (F).

To collect the sperm, three spawning males were transferred to a separate 30-L capacity circular container. Each breeding female was quickly transferred to a separate 100-L capacity circular container at the first sign of egg release. No aeration was provided on the circular container as this may cause disturbance to spawning. After spawning, the spent females were removed and 5 ml of water containing sperm taken from a separate container with spawning males was added at each tank with eggs. All 55 breeders were released in the wild the after the induced spawning.

Estimating the productivity of captive trochus

The number eggs produced by each female was estimated by volumetric method, performed by taking the average number of eggs from five vial samples and multiply it with the volume of water in the container. The numbers of fertilised eggs and hatched larvae were estimated after 12 and 36 h from the time of adding the sperm using the same method. Fertilisation rates were expressed as a percentage of fertilised and total number of eggs per female that spawned. Hatching rate was expressed as the number of hatched larvae as percentage of the number of fertilised eggs.

Data analyses

The survival analysis was based only on the total initial number of newly collected breeders (n=106) instead of 117 individuals because the 11 old spawners of WPU-SF died within a few days of release, possibly due to weeklong sea turbulence which coincided with the time of stocking. The cumulative recovery rates were expressed as the percentage of live shells recovered from the tanks and total initial stock. Potential cumulative survival rates were obtained by dividing the monthly total number of live shells recovered plus escapement with the total initial stock that month x 100. Cumulative mortality rates were expressed as percentage of empty shells recovered every month from the intertidal tank and the total initial number of stock that month. Average monthly growth rates were obtained by subtracting the initial size from the size at recapture, or subtracting the size at previous recapture from the size during the succeeding recapture. Only the trochus with complete record (n = 48) for 11 months was included in computing the mean monthly diameter and weight. This was obtained by using the repeated measure command under the generalised linear model protocol of the SPSS software version 16. Pearson correlation in SPSS version 16 was used to determine the relationships of total rainfall, salinity and water temperature on the growth of trochus (Field 2009). The monthly total rainfall (mm) was obtained from a local weather station (Philippine Atmospheric, Geophysical and Astronomical Services Administration) located about 50 km straight from the study site.

RESULTS

Growth of trochus reared in intertidal tanks

Monthly average diameter was increasing but growth increment was very low, averaging only 2.2 mm (Range: 0.0 – 18.7 mm; sd: 3.7 mm, n = 48) increase in 11 months for trochus with initial diameter ranging between 61.5 to 100.6 mm. Five smaller individuals with a size range of 61.5 – 71.0 mm at the start of the study had faster growth with a mean (± sd) increase of 12.6 ±3.7 mm in 11 months. Larger individuals with initial size of 73.3 – 84.7 mm have much lower mean (± sd) diameter increment of 1.8 ± 1.6 mm in 11 months.

Average monthly diameter increments were higher (range: 0.26 – 0.45 mm mo^{-1}) between November and December, and between July and August compared with increments in other months of the year (range: 0.08 – 0.19 mm mo^{-1}) (**Figure 4.2A**). The trend in weight was increasing towards the end of the study but was characterised by negative growth during the months of October, March, May and August (**Figure 4.2B**). The lowest average weight increment in August (-0.88 g) coincided with the highest diameter increment (0.45 mm).

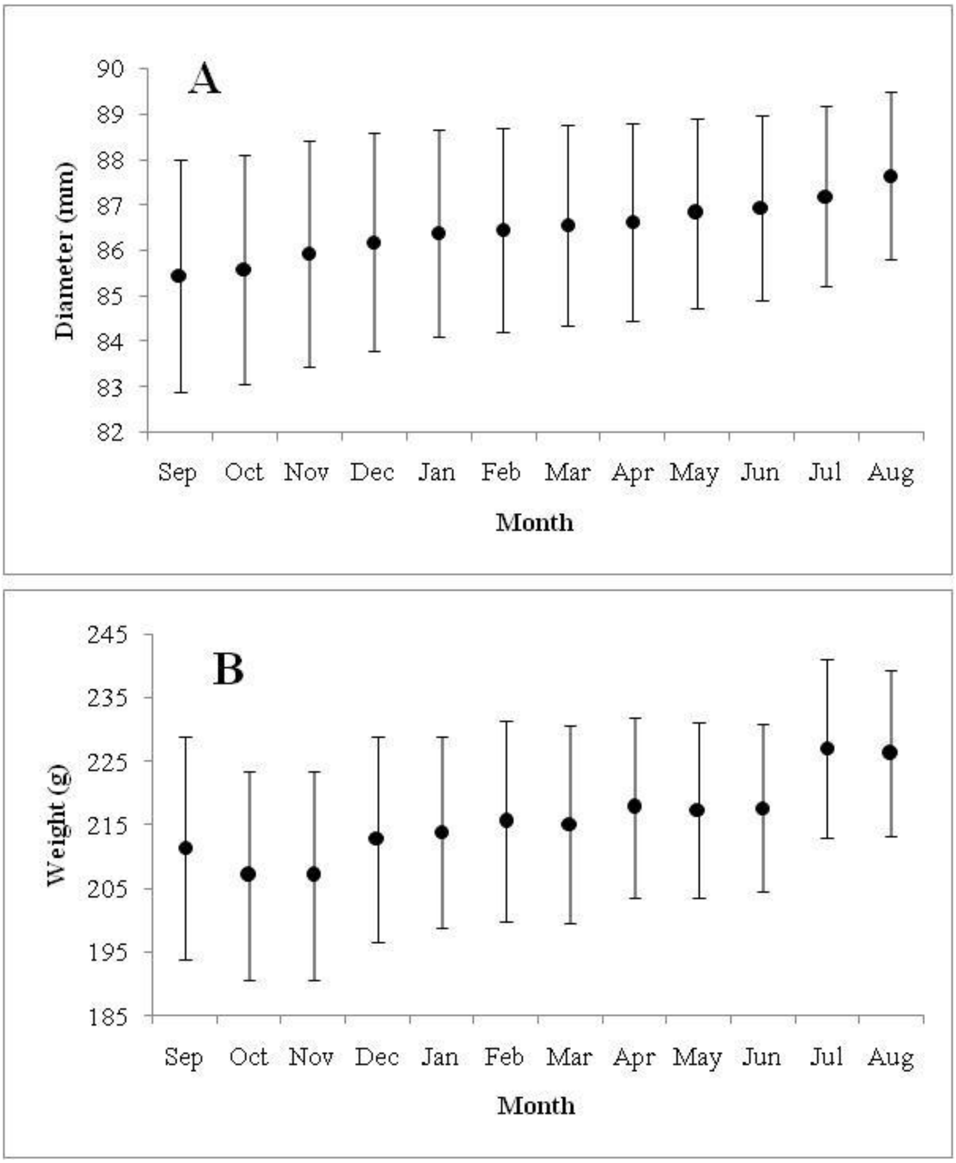

Figure 4.2. Mean monthly (A) diameter (mm mo^{-1}) and (B) weight (g mo^{-1}) of trochus held captive at intertidal tanks for 11 months starting from September 2009 until August 2010. The average size (± 95 % CI) was computed from 48 individuals having a complete record of diameter and weight for the duration of the study.

Correlation analysis on monthly average diameter and weight were significant ($r = 0.911$, $P < 0.001$, n = 12). However, the average monthly increment in diameter was not related to weight increment (r = 0.06, P > 0.05, n = 11). Rainfall, salinity and water temperature has no significant relationship with diameter and weight increments (**Table 4.1**). However, rainfall appeared to have a contrasting relationship with growth for having a moderately high positive relationship with diameter increment, but with a low negative correlation with weight increment (**Table 4.1, Figure 4.3**).

Table 4.1. Relationships between water parameters and growth increments for trochus breeders held captive for 11 months in intertidal tanks.

Correlations

		Diameter Increment (mm/mo)	Weight Increment (g/mo)	Total Rainfall (mm)	Salinity (ppt)	Water Temperature (oC)
Diameter Increment (mm/mo)	Pearson Correlation	1	.101	.578	-.025	.059
	Sig. (2-tailed)		.768	.062	.942	.862
	N	11	11	11	11	11
Weight Increment (g/mo)	Pearson Correlation	.101	1	-.303	.487	.318
	Sig. (2-tailed)	.768		.365	.129	.341
	N	11	11	11	11	11
Total Rainfall (mm)	Pearson Correlation	.578	-.303	1	-.575	.222
	Sig. (2-tailed)	.062	.365		.064	.511
	N	11	11	11	11	11
Salinity (ppt)	Pearson Correlation	-.025	.487	-.575	1	.089
	Sig. (2-tailed)	.942	.129	.064		.794
	N	11	11	11	11	11
Water Temperature (oC)	Pearson Correlation	.059	.318	.222	.089	1
	Sig. (2-tailed)	.862	.341	.511	.794	
	N	11	11	11	11	11

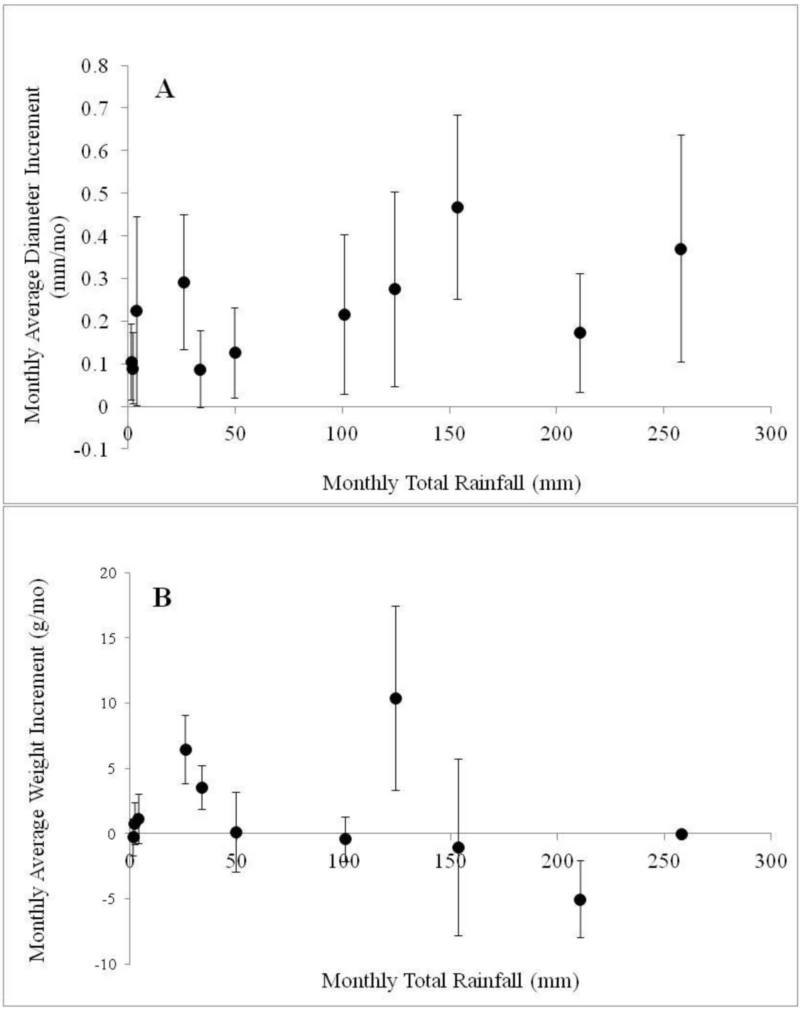

Figure 4.3. Relationships between total monthly rainfall (mm) and average monthly diameter (A) and weight (B) increments for trochus raised in intertidal tanks for 11 months. Error bars are 95 % confidence intervals.

Survival rates of captive trochus in intertidal tanks

The survival rate of broodstock based on recovered live individuals after 11 months in captivity in intertidal tanks was about 51.9 %. Twelve percent of the lost shells were found alive. If all lost individuals (28.3 %) were alive, survival rate could be as high as 80.2 %. A total of seven (6.6 %) live individuals inside the tanks were missed in four sampling events. Mortality rate based on recovered dead shells was 20 %. Nearly 60 % of the tags were damaged after 11 months.

However, because of pencil marking and another small tag, the shells were easily recognised every sampling event (**Figure 4.4**).

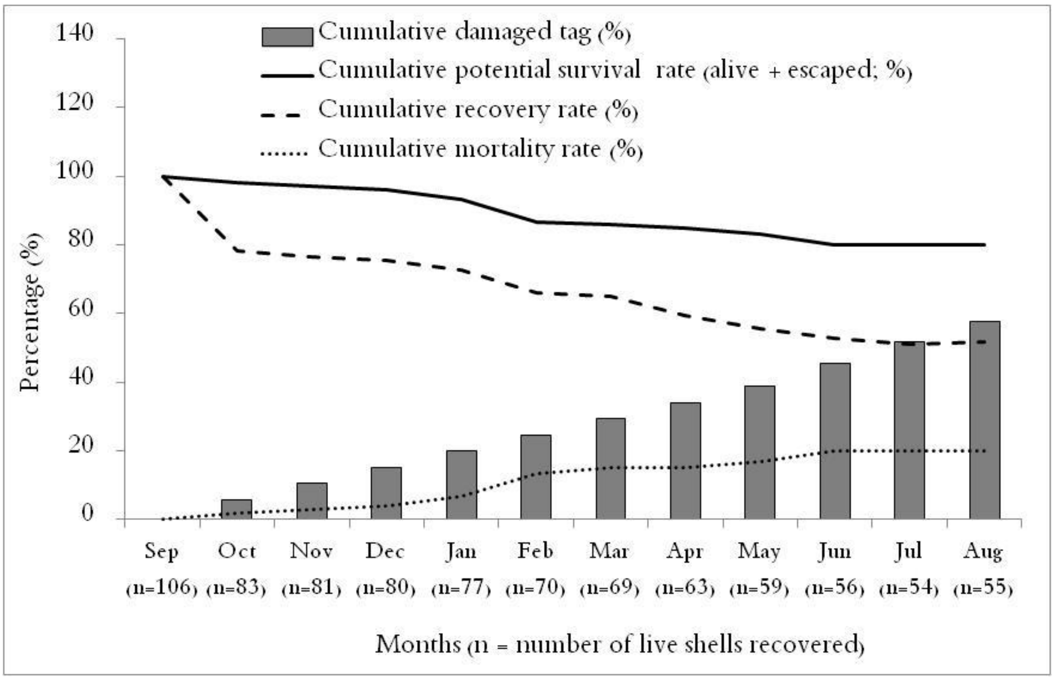

Figure 4.4. Cumulative recovery rates (dashed line), potential survival rates (solid line), mortality rate (dotted line) and rate of tag loss (histogram) for trochus held captive in intertidal tanks for 11 months. In July, one sample was uncollected from the tank and this was recaptured in August. In total, seven live individuals were missed inside the tanks in four sampling occasions.

Spawning, fecundity and fertilisation and hatching rates

After 11 months in captivity, a total of 55 live individuals were recovered from the intertidal tanks and all these shells were induced to spawn, of which, 23 (42 %) individuals have spawned successfully. The number of male trochus (19 ind) that spawned was about 5 times higher than females (4 ind). The average (± sd) shell diameter of breeding males (87.0 ± 5.7 mm) and females (87.2 ± 5.7 mm) were almost similar. The number of spawned eggs ranged from 81 000 to 435 000 with an average (± se) of 184 000 (± 84 000) eggs per female (**Figure 4.5**). Fertilisation rate averaged at 87 % (95 % CI: 77.3 – 96.7 %) of total number of eggs and hatching rate was 81 % (95 % CI: 73.8 – 88.2 %) of fertilised eggs respectively. Total number of larvae produced was nearly 0.6 million out of 0.737 million eggs.

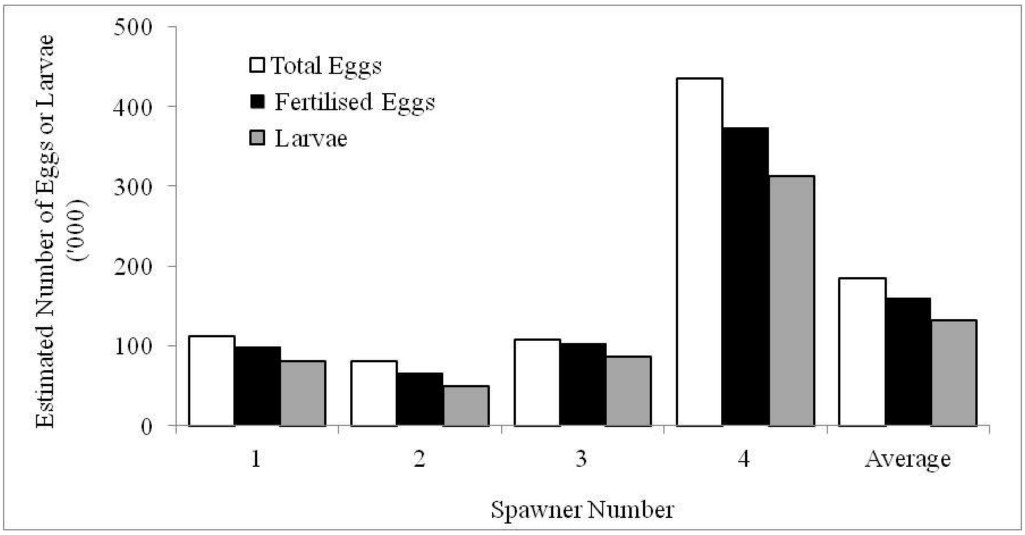

Figure 4.5. Reproductive output of four female trochus breeders that responded to induced spawning after 11 months of captive rearing in intertidal tanks.

DISCUSSION

Growth and survival of trochus held captive in intertidal tanks

The average growth increment (2.2 mm in 11 months) of breeders held captive in intertidal tanks was much slower than in the wild. Trochus > 80 mm in diameter can grow at least 10 mm yr^{-1} (Smith 1987; Nash 1993; Lemouellic and Chauvet 2008), much higher than the average growth of captive trochus in this study. This extremely slow growth of trochus could have been due to limited food supply. Encrusting algae inside the tanks could have been consumed shortly after stocking resulting to a decline in weight during the first month (September – October) which was corrected by adding of coconut fronds starting October and onwards (**Figure 4.2B**). Captive 4 – 7 mm trochus juveniles (n = 20 000) can graze out a 6.5 m^2 of cultured diatoms on tanks surface in a week (Dwiono et al. 1997). Food shortage can slow down the growth of juveniles in tanks (Heslinga and Hillmann 1981) and in another experiment, Hoang et al. (2008) have enhanced the growth of trochus juveniles by using a variety of foods. Trochus are known to feed on a variety of seaweeds (Lambrinidis et al. 1997), plants and animals (Rao 1937; Soekendarsi et al. 1998). Indonesian fishermen feed trochus kept in enclosures with papaya or banana stems and animal skins (Burhanuddin 1997). Juvenile trochus in indoor tanks provided with coconut leaves as substrate developed a reddish streak coloration (**Chapter 5**) suggesting the intake of plant materials.

Salinity and temperature were not significantly related with growth increment. Seasonal growth in the wild has only been reported in the southern and northern parts of its range with contradicting results, but not in tropical areas like the Andaman Islands (Nash 1993) and Guam (Smith 1987). The moderately high positive correlation between monthly total rainfall and monthly average diameter increment (r = 0.58), although not significant (P > 0.05), could have been an effect of high organic particles drained into the sea in times of heavy rain that trochus can feed on. The lowest diameter increments ($\leq$ 0.09 mm mo^{-1}) between February and April coincided with the lowest average total rainfall (10.2 mm; range: 1.3 – 33.5 mm mo^{-1}) that fell between December and March. The growth of trochus juveniles in sea cages that graze on organic particles was faster than those maintained in indoor tanks (**Chapter 5**). The contradicting trends of diameter and weight increments in relation to total rainfall (**Figure 4.3**) could be an indication of natural spawning in intertidal tanks. Natural spawning can occur in indoor aquaria (Nash 1985) and spawning may occur year round in the wild (Hahn 1993; Gimin and Lee 1997c).

The high survival rate (51.9 – 80.2 %) of trochus reared for 11 months in intertidal tanks was comparable with the survival rates of 46 – 77 % (Nash 1993) and 81 – 92 % (**Chapter 6**) in the wild. After our study in August 2010, the

WPU-SF adapted the idea of keeping the breeders in intertidal tanks after the induced spawning where they obtained 80 % survival instead of zero survival of breeders used in earlier induced spawning (Avillanosa et al. 2010). The cause of trochus mortality in intertidal tanks is not known, but recovered dead shells were unbroken or unoccupied by hermit crabs, suggesting stress related mortalities. Lost trochus hid under crevices or had moved towards the deeper part of the reef, thus the 51.9 % survival rate based on recovered live shells after 11 months is a conservative estimate.

The growth and survival of trochus at intertidal concrete tanks could be comparable to those in the wild when ample supply of food is provided. At the same time, if spawning naturally occurs in such captive condition, it is likely that fertilisation rates during spawning is much higher compared to when trochus are at low densities in the wild. There is no known information on fertilisation and hatching rates of trochus in the wild but a decline in fertilisation success with increasing separation distance between spawners had been reported for abalone *Haliotis laevigata* (Babcock and Keesing 1999). Allee effects coupled with overfishing had been pointed out to cause the extinction of a giant clam species in some tropical island countries in the Pacific (Hobday et al. 2001).

Spawning and fecundity of trochus held captive in intertidal tanks

In this study, the percentage of breeding males (34 %) and females (7 %) in relation to the total number of breeders was higher than the average percentage of successfully induced males (13 %) and females (4 %) at IMDC between 1999 and 2006 (**Chapter 3**). Lower percentages of breeding individuals (23 % males and 5 % females) were also experienced by WPU-SF (Gonzales et al. 2009). Hahn (1993), found that female trochus have a reproductive cycle of 8 months and that 12 % of the total population must spawn every new moon of each lunar month. However, all the reported percentages of breeding females were low and only spawning rates in our study are higher but were still only half of Hahn's

(2003) estimated monthly percentage of naturally breeding females. Factors that may have influenced these could include: 1) the number of breeders that were ready to spawn at the time of induced breeding; 2) stress due to improper handling during collection. At harvesting, collected trochus were left exposed to the sun or rain on a bamboo flooring of the boat, or placed on the hull of the boat which may contain some water contaminated with fuel (pers. obs.). Another observation of the IMDC staff was the high probability of non response to induce spawning when the shells were kept by fishers overnight (Cayaon, pers. comm., **Chapter 3**). Nash (1989) suggested that to avoid stress and induce spawning in the field, trochus must not be kept overnight by the fishermen; 3) the sex ratio of captured trochus favours males because of differential behaviour after spawning because females tend to hide after spawning (Nash 1985; Nash 1993); and 4) possibly, such a high percentage of breeding males is a reproduction strategy to ensure high fertilisation rates. Nash (1985) noted that unlike female trochus that spawned following the lunar cycle, male trochus spawned with no recognisable periodicity. Identifying the sex and conditions of eggs of breeders prior to induced spawning can help explain why fewer numbers of females are responding to induced spawning than the male trochus.

The average fecundity (180 000 eggs) in this study was 30 % lower compared to a value of 270 000 eggs per breeder with a mean (± sd) size of (87.6 ± 15.9 mm) in Palau (Kitalong and Orak 1997). However, 62 % of those breeding females in Palau had only released < 200 000 eggs. The mean (± sd) size (83.3 ± 16.0 mm) and fecundity (116 000 ± 52 000 eggs) of those females were relatively low compared to the reproductive output of trochus in this study. The reported average number of eggs per spawner were much higher in IMDC (400 000 eggs per female). There was no record on the size of each breeding female at IMDC but the mean size (computed from mean size per induced breeding) of all spawners was 93.0 ± 9.8 mm (**Chapter 3**), much larger than those of Kitalong and Orak (1997).

The absence of relationship between the size of the females and the number of eggs they released during the induced spawning could be related to partial spawning as a result of stress or disturbance to spawning females. Several authors have established a direct relationship between the size of female and numbers of eggs contained within the ovary of individual females (Heslinga 1981b; Nash 1985; Bour 1989), but there was no clear pattern between the size of the spawners and the numbers of eggs they naturally released in captivity (Nash 1985; Kitalong and Orak 1997; Gonzales et al. 2009) (**Figure 4.6**). At IMDC, incidences of failure spawning (no response at all) and discontinued spawning were not uncommon (Cayaon pers. comm., **see Chapter 3**). The reasons are unknown but we presume that this may be related to handling and stress from the time of collection until the induced spawning. If partial spawning occurs in the hatchery, the reproductive output of the females are not maximised by induced spawning. Hahn (1993) concluded that trochus are complete spawners because only one cohort of gametes were developing at a time in each shell, while Pradina (1997) postulated that trochus are capable of either complete or partial spawning, with the higher tendency to spawn partially as manifested by the variation of gonad development among the female shells.

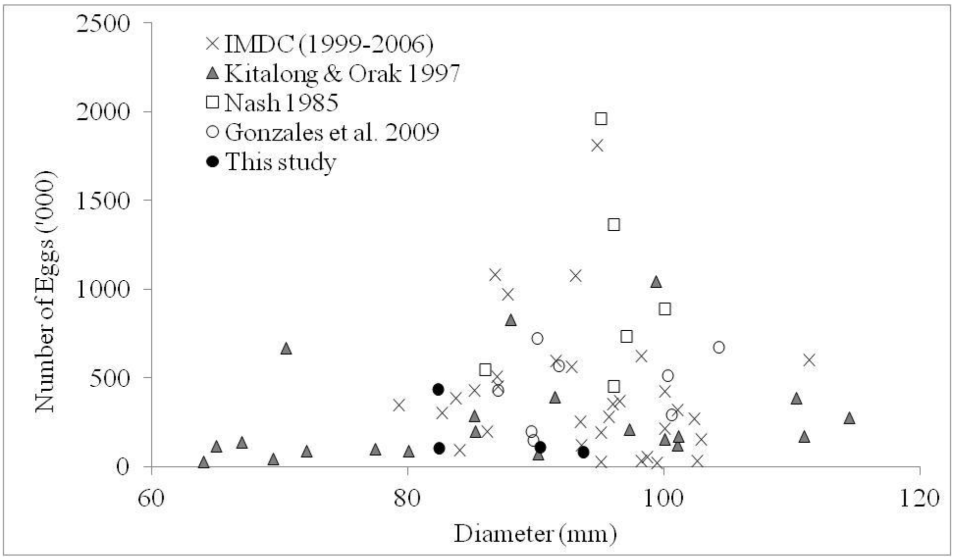

Figure 4.6. Realised fecundities of trochus held captive in intertidal tanks for 11 months compared with other studies.

The average fertilisation (87 %) and hatching (81 %) rates in this study were higher compared to that of Gonzales et al. (2009) which were 58.91 % and 65.97 %, and that of IMDC (80 % and 60 %, **see Chapter 3**). Hatching rates were much lower at SEAFDEC-AQD where veligers only comprised 10 – 36 % of the total number of eggs (Gapasin et al. 2002). Kitalong and Orak (1997) reported fertilisation and hatching rates of 82 % and 74 % which were also a little lower than in the recent study. Dwiono et al. (1997) relate the hatchability of eggs to time at which females responded to spawning experiment. Eggs obtained from females that spawned on the first or second day of the spawning experiment had higher hatching rates than the eggs produced by females which responded 10 days later.

Implications for management

The artificial propagation of trochus and the release of hatchery-produced juveniles had been viewed as one of the management options to revive depleted trochus populations (Heslinga 1981a; Heslinga and Hillmann 1981; Isa et al. 1997b; Lee 1997) , but the practice of keeping the breeders in indoor tanks after the breeding process which reduces their reproductive output and shortens their life span, could possibly bring more harm than good to trochus populations.

Hatchery produced juveniles can have zero (Isa et al. 1997b) or very low (Castell 1993; Crowe et al. 2002) survival rates in the wild, thus only small numbers could reach sexual maturity and begin to reproduce. Such numbers might not even be as many as the natural recruits produced by those parent shells if left to reproduce in the wild. This is an intuitive reasoning but since it takes about 1 – 2.5 years to raise trochus from hatching until sexual maturity (Heslinga 1981a; Lee 1997), that period would have allowed those breeders to produce millions of eggs and potential recruits when they were returned in the wild.

The potential reproductive output of trochus when allowed to breed naturally in intertidal tanks could be far higher than offspring produced through induced breeding. Assuming that 30 % of the captured 171 breeders of WPU-SF were females because of differential reproductive behaviour (Nash 1993) and with a reproductive cycle of 7 – 8 months (Hahn 1993), this means that at least each female breeds at least three times in two years. Using a conservative average realised fecundity of 200 000 eggs per spawning, then at least 30 million eggs were naturally released in two years. If the reproductive output is based on the potential fecundity, where an ovary of 87 mm shell may contain at least 800 000 eggs (Bour 1989) , the potential reproductive output in two years (of the assumed 30 % female of the 171 breeders) would be more than a 100 million eggs. By contrast, the WPU-SF had induced to spawn a total of 171 trochus in five separate occasions between April and May 2009 with nine breeders only spawned a total of 3.5 million eggs (Gonzales et al. 2009) or 3.6 million eggs as reported by Avillanosa et al. (2010). The IMDC had produced in about five years a total 48.38 million eggs out of 2 348 breeders. Of which, only 1.08 million juveniles (3 – 36 mm) and 5.7 million larvae were released but there were no harvest of released trochus possibly because of high mortalities of the seeded juveniles and uncontrolled gleaning at the sites of release (**Chapter 3**).

Given the above-mentioned scenario, it could be more advantageous to keep a small number of trochus in intertidal tanks in promoting natural recruits than engaging in laborious and costly hatchery propagation. Trochus in their natural habitat spawn throughout the year (Heslinga 1981b; Hahn 1993; Gimin and Lee 1997c). They also spawn naturally under indoor conditions (Heslinga 1981a; Heslinga and Hillmann 1981; Nash 1985) and more so when kept under favourable conditions in intertidal tanks or in sea cages. Fertilisation rate among broadcast spawners increases with density per square meter (Lundquist and Botsford 2004). And a decrease in fertilisation rate with increasing distance between spawners had been documented in abalone species (Babcock and Keesing 1999). Keeping therefore of sexually identified breeding trochus in intertidal tanks could be an effective stock enhancement tool when combined with good husbandry and effective protection of the naturally occurring populations. The use of grow-out sea cages facility in stock enhancement had been found cost effective for the sea urchin (Juinio-Meñez et al. 2008).

The current intertidal tank set up requires further modifications because it is frequently exposed during low tide. There is a need to improve the tank's cover to facilitate easy introduction of rocks grown with seaweeds. To avoid excessive exposure of the shells to sunlight, future deployment of concrete tanks should be in shallow subtidal areas. Tank construction and deployment could be expensive, that the use of light and cheap aluminium cages set on the leeward side of the reef may be a better alternative. While keeping such breeders to naturally reproduce in intertidal tanks or cages is substantially cheaper and could be a more effective tool in stock enhancement than the seeding of artificially produced juveniles, this method requires periodic monitoring and good husbandry making it less advantageous compared to having trochus in uncaged condition. The rearing of trochus in intertidal tanks for stock enhancement should be limited to projects of government institutions to avoid its use as cover up for illegal trade. Lucrative turtle farms in China are considered a major threat to its diverse turtle fauna because these farms are the primary purchaser of wild-caught turtles which can lead to the extinction of wild populations (Shi et al. 2007).

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Chapter 5

INDOOR AND DEEP SUBTIDAL INTERMEDIATE CULTURE OF *Trochus niloticus* FOR RESTOCKING

Roger G. Dolorosa, Alastair Grant, Jennifer A. Gill,
Arlene L. Avillanosa and Benjamin J. Gonzales

ABSTRACT

The high demand for shells of the large reef associated gastropod *Trochus niloticus* in the manufacture of mother-of-pearl buttons has resulted in a widespread decline of its population. As a consequence, juvenile mass production and restocking has been practiced as one of the many conservation measures. Trochus has long been successfully bred in captivity, but culturing of juveniles until ready for release is faced with many problems. One is shortage of natural food. Terrestrial plants have been traditionally used by fishermen as food in keeping wild trochus juveniles but their potential use in intermediate culture of trochus has not been evaluated. We conducted four growth trials for 60 – 120 days, rearing hatchery-produced juveniles (10 – 28 mm shell diameter) at different stocking densities in indoor tanks and sea cages, with coconut leaves as the main or an additional substrate. The average growth rates of 4.4 mm mo^{-1} (95 % CI 4.0 – 4.7 mm mo^{-1}) for all stocking densities in one of the growth trials involving the use of small cages deployed at 5 – 6 m on the reef slope were comparable to the growth in the wild. This growth rate was three times higher compared with growth trial in large metal cages on the reef slope, and 2 to 23 times higher than in two indoor conditions such as the use of wooden tanks and small cages set in a tank. Survival rates were as high as 99 %. Incidence of escape in subtidal cages was low except when some cages were blown by strong waves. The results indicate that trochus juveniles can be successfully cultured at high density in subtidal cages with coconut leaves as substrate.

Keywords cage culture, indoor tanks, growth rate, stocking density, survival

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INTRODUCTION

The reef gastropod *Trochus niloticus,* also known as trochus, is an important source of food and revenue for many countries in the tropical Pacific (Heslinga and Hillmann 1981; Nash 1985; Nash 1993; Hahn 2000). Their limited distribution and increasing demand from the shell trade stimulated the introduction of wild trochus to many offshore reefs and islands, expanding its distribution range (Gillett 1993), and trochus has become an important source of revenue in some of these areas (Passfield 1997; Ponia et al. 1997; Zoutendyk 1997). By contrast, many trochus populations in its native range have been threatened due to overharvesting (Nash 1993; Dwiono et al. 1997; Smith et al. 2002; Hoang et al. 2007; Dolorosa et al. 2010), giving rise to an interest in understanding its biology, so that these populations can be more effectively conserved. While mass production of trochus has been successfully carried out (Heslinga 1981; Heslinga and Hillmann 1981; Isa et al. 1997; Lee 1997), the high susceptibility of small juveniles to predation when released into the wild (Castell 1993; Chauvet et al. 1997; Isa et al. 1997; Crowe et al. 2002) requires extended rearing until a suitable release size of > 30 mm (Isa et al. 1997; Purcell et al. 2004) is attained. Extended rearing in hatcheries can be costly and limits production (Heslinga 1981; Isa et al. 1997), so the use of sea cages for intermediate rearing has been practiced, but with varying success.

Purcell (2001) successfully raised 15 mm trochus to a size of 33 mm in 11 months at 30 ind m^{-2} in cages in intertidal areas. However, these cages were deployed in an unsheltered area, which was susceptible to shifting sand, piling up of substrate in one corner of the cage, food shortage and destruction by a cyclone. Amos and Purcell (2003) also successfully raised 15 – 30 mm juveniles to a size of 40 – 50 mm at 30 ind m^{-2} in 9 months of rearing in reef-based and floating cages. They pointed out the importance of cage design to minimise escape and mortality of stock which occurred excessively in reef-based cages, because those that escaped were unlikely to survive unless they were more than 30 mm in size (Isa et al. 1997). Purcell (2001) suggested an ideal density of 30 ind m^{-2} in sea cages, but Hoang et al. (2007) recommended a gradual density reduction as shell size increases (10 – 22 mm at 100 ind m^{-2}; 25 – 40 mm at < 50 ind m^{-2} and 40 – 50 mm at < 10 ind m^{-2}).

Factors affecting trochus growth include density (Amos and Purcell 2003; Hoang et al. 2007), type of locality (Nash 1993; Castell 1997), substrate (Gimin and Lee 1997b), food availability (Heslinga 1981; Purcell 2001; Hoang et al. 2007; Hoang et al. 2008), and environmental parameters (Nash 1993; Gimin and Lee 1997a). Season has been reported to affect the growth of trochus in the northern and southernmost part of its distribution but no seasonal growth variation was reported in Palau or India (Nash 1985; Nash 1993).

Trochus are known to feed on a wide variety of plant and animal taxa including some detritus (Soekendarsi et al. 1998). They exhibit food selectivity (Isa et al. 1997), generally preferring soft filamentous over the leathery brown algae (Lambrinidis et al. 1997a; Clarke et al. 2003). Aside from naturally occurring food in the wild, feeding juveniles at night are attracted to papaya and banana stems (Burhanuddin 1997) and on coconut leaves drifted on intertidal areas (B.J. Gonzales *pers. comm.*). In Indonesia, wild trochus held in enclosures have been fed with plant materials, animal skin and other organic matters (Burhanuddin 1997). The substrate most commonly used in sea cages is algae-covered coral rocks (Amos and Purcell 2003; Hoang et al. 2007) needing to be replaced once the food are over grazed. However, the process of replacement would require more labour, produce a danger of introducing predatory species into the cage, and increase difficulty in recapturing or sampling of shells.

Sea cages are vulnerable to wave action (Purcell 2001) but when the people in the village are interested in cage-rearing of trochus juveniles for reseeding, an option could be to deploy the cages in sheltered areas or on the deep reef slope which is less affected by wave conditions.

In this study, we explored the growth and survival of hatchery produced trochus juveniles at different stocking densities both in indoor tanks and cages on the reef slope with the use of coconut leaves as supplemental or main substrate. Specifically, we conducted a series of experiments to (a) test the effectiveness of two cage designs deployed on the reef slope; (b) determine the growth and survival of trochus at different stocking densities at each experiment conducted either in indoor tanks or on cages on the reef with coconut leaves as substrate.

METHODS

Source of juvenile *Trochus niloticus* used in growth and survival experiments

Trochus niloticus juveniles produced by induced spawning between April and May 2009 at the Western Philippines University – Samong Farm (WPU-SF) in Binduyan, Puerto Princesa City, Palawan, Philippines (10^{o}1.379' N and 119^{o}6.337'E) were used in the growth experiments.

Tagging procedure for *Trochus niloticus* juveniles

Trochus suitable for tagging (> 10 mm) were collected from the indoor grow-out tanks and randomly assigned to treatments for each experiment. Each shell received a 6 mm pre-numbered TZ dymo tag using the following steps:

1. Wipe the shell's surface with tissue paper; allow the shell to dry for few minutes.
2. Drop a small amount of loctite glue (Mighty Bond) onto the dried area.
3. Carefully place the tag on the portion with glue.
4. Allow the glue to dry for few minutes before reapplying on and around the tag.

5. Allow the glue to dry for about an hour while keeping the shells out of the sun and in an upright position over a wet cloth to minimize stress and avoid desiccation.

Design of juvenile *Trochus niloticus* growth and survival experiments

1. **Experiment 1** (E1): Nine net cages (0.25 x 0.25 x 0.15 m) made of 5 mm round metal frame wrapped with 0.5 cm meshed net were used. Each cage had a plastic corrugated plate fixed at the bottom to support a small coral rock sinker. Each cage received four pieces of brown coconut leaflets as substrate, enough to cover the floor area before setting it in an indoor tank (**Figure 5.1A**).

2. **Experiment 2** (E2): Nine net cages of the same design used in E1 were held together in a 0.75 x 0.75 m wooden frame (**Figure 5.1B**) prior to deployment on a 5 – 6 m deep reef slope. Each cage was provided with four pieces of coconut leaflets as substrate, roughly enough to cover the basal area of the cage.

3. **Experiment 3** (E3). Eight wooden tanks (2 x 1.65 x 0.6 m) housed in a semi indoor shelter were used. At the start of the experiment, each tank received 8 sets of corrugated plates (each set is composed of 20 sheets of 0.4 x 0.78 m plates) grown with filamentous green algae. Dried coconut fronds were added as substrate on the 30th day when most algae on corrugated sheets were overgrazed. At the end of the 60-day experiment, 97 % were recovered alive and were transferred to eight steel cages deployed at the reef slope and is referred as Experiment 4 (**Figure 5.1 C & D**).

Figure 5.1. Small cage used in E1 (A); a set of small cages in E2 (B); the wooden tanks used in E3 (C); tagged trochus crawling on coconut leaves placed in wooden tanks (D); tagged trochus having a distinct reddish mark as an effect of coconut substrate (E); preparing the large cages (E4) for deployment (F); deployment of a large cage on the reef slope (G); trochus feeding on thick organic particles that settled on coconut leaves (H).

4. **Experiment 4** (E4). Eight steel cages (1 x 1 x 0.3 m) with 2.5 cm angle bar frames and three layered walls: steel matting (5 cm mesh size), plastic net (2.5 cm mesh size) and fish net (0.5 cm mesh size) were used. For easy replacement of substrate and collection of experimental animals, the central upper side of each cage was provided with a 50 x 50 cm window (**Figure 5.1F**). Two small round timbers were tied along the bottom sides of each cage, to keep it off the ground and/or to avoid being buried under sand. Dried coconut leaves as substrate were firmly tied to each cage, covering about 80 – 90 % of the cage's floor area. The steel cages were deployed near the E2 (**Figure 5.1G**).

Because about 3 % of the shells were not recovered during the final sampling from E3, those were replaced to complete the number of shells at each treatment in E4. This resulted in a slight increase (up to 0.05 mm) in the average size at each treatment. The cages stocked with these juveniles were haphazardly deployed on 5 – 6 m deep reef slope (**Figure 5.1G**).

Duration of the study, stock management, sampling and data analyses

The duration of four growth experiments varied from 60 – 120 days. The first indoor growth experiment (E1) was conducted for 120 days (October 2009 – February 2010), while E3 was conducted for 60 days (March – May). Both outdoor growth experiments (E2 and E4) were conducted for 90 days (May – August 2010). The ages of trochus at the start of the experiment ranged between 5 – 12 months old. The stocking density per experiment increases with Treatment number, with T1 having the lowest density and T3 or T4 with the highest density. Both E1 and E2 had three treatments (T) with three replicates, while both E3 and E4 had four treatments and two replicates (**Table 5.1**).

Table 5.1. Densities (ind m^{-2}), mean sizes (mm) and ages (mo) of *T. niloticus* at the start of each experiment. T = Treatment.

Experiment code (no. of replicates)	Month and Duration (days)	T	Density		Mean size ± se (mm)	Starting Age (mo)
			ind per cage or tank	ind m^{-2}		
E1 (3)	Oct - Feb (120 days)	1	2	32	16.43 (±0.73)	5
		2	4	64	15.66 (±0.52)	
		3	6	96	15.58 (±0.42)	
Mean (n=36)					15.75 (±0.29)	
E2 (3)	May - Aug (90 days)	1	2	32	12.68 (±0.38)	12
		2	4	64	11.77 (±0.27)	
		3	6	96	11.67 (±0.22)	
Mean (n=36)					11.87 (±0.16)	
E3 (2)	March - May (60 days)	1	50	15	15.48 (±0.16)	10
		2	100	30	14.77 (±0.11)	
		3	150	45	14.38 (±0.09)	
		4	200	60	14.45 (±0.08)	
Mean (n=1000)					14.60 (±0.05)	
E4 (2)	May - Aug (90 days)	1	50	50	20.19 (±0.24)	12
		2	100	100	20.20 (±0.17)	
		3	150	150	18.36 (±0.14)	
		4	200	200	18.67 (±0.12)	
Mean (n=1000)					19.04 (±0.08)	

The diameters of the shells at the start of the study and every after 30 days were measured with sliding callipers to the nearest 0.1 mm. The number of dead shells was recorded but these were not measured as the time of death was not known. The dried coconut leaves used as substrate were replaced every month to avoid decay and fouling of the water especially for indoor tank experiments. For E3, the coconut leaves were only introduced during the second month when most algae growing on corrugated plates were overgrazed.

Experiments under indoor conditions (E1 and E3) received constant aeration except during power black outs which frequently occurred in times of bad weather. Water exchange was conducted every two or four days by flow through system using sand filtered sea water, replacing approximately 90 % of the water. Total water exchange for E3 was conducted every monthly sampling period.

Trochus with lost tags were removed in the analyses of growth but were included in survival analyses. The growth among treatments per experiment was analysed separately using analysis of covariance, with the initial size as covariate and sizes at succeeding months as dependent variables and treatment as fixed factors. Homogeneity of variance of the initial size (covariate) between treatments at each experiment was determined with Levene's test. In cases where this test indicated significant differences among the variances, the ratio between the largest and the smallest variance was compared with the critical values to double check the homogeneity of variance to justify the use of analysis of covariance (Field 2009).

Results of these tests revealed no significant variation among variances of covariates in each experiment ($P > 0.05$). *Post hoc* tests were conducted using Bonferroni correction to exactly determine the treatment that significantly differs from the other. Cumulative recovery rates (RR) of live shells were calculated by dividing the total number of live shells recovered per month by the total initial stock at each treatment. Cumulative rate of losses of shells were computed by adding the numbers of recovered dead shells and number of lost shells (which have crawled up the wooden tank or those which able to escape from cages on the reef) and dividing it with the total initial stock at each treatment. Comparison of RR of live shells was conducted separately at each experiment using one-way analysis of variance and Scheffé post hoc tests (Field 2009) to determine the variation in survival at different densities. All comparisons were conducted using the SPSS software version 16.

RESULTS

In comparing the sizes (mm) of trochus at different densities per experiment, significant differences were noted on the final sampling of three (E2, E3, E4) growth experiments, but not in E1. The survival rates of trochus at different stocking densities per experiment did not significantly differ among treatments per growth experiment.

Experiment 1 – Small cages set in indoor tanks

Trochus kept in this design for 120 days had extremely slow growth rates, and there was no significant variation in diameter among the three stocking densities at the end of the study ($F_{(3, 31)} = 0.17$, $P = 0.84$, **Table 5.2**, **Figures 5.2 & 5.6**). Recovery rates of live shells, however, were high (96 – 100 %) after 120 days and did not vary significantly among treatments ($F_{(2, 9)} = 1.0$, $P = 0.40$, **Table 5.2**). There was no incidence of escape from this set up (**Figure 5.7**).

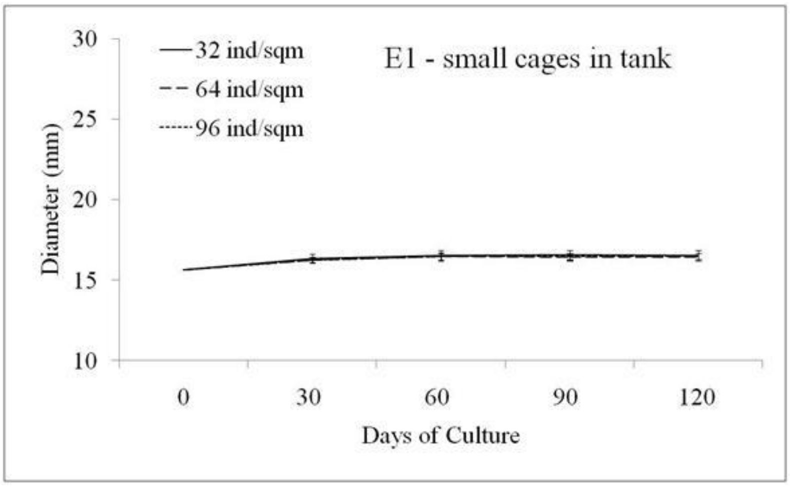

Figure 5.2. Monthly mean diameter (mm) of *T. niloticus* raised at different stocking densities in E1 (small cages set in indoor tank). The error bars are 95 % confidence intervals. The initial size at zero day of culture is the average size of all trochus used in the experiment.

Experiment 2 – Small cages set on a reef slope

The mean sizes of trochus at different densities or treatments did not significantly differ in the first 60 days ($F_{(2, 29)}$ = 0.083, P = 0.92, **Table 5.2, Figures 5.3 & 5.6**). However, on the 90th day, there were significant variation on sizes among treatments ($F_{(2, 21)}$ = 4.45, P = 0.024, **Table 5.2**). The combined average (± se) growth rates on the 30th and 60th day of rearing were 4.8 ± 0.19 mm mo^{-1} and 5.7 ± 0.14 mm mo^{-1} respectively. The mean (± se) growth rates (1.9 ± 0.17 mm mo^{-1}) for all treatments on the 90th day have declined by > 60 percent relative to the mean growth rate on the 60th day, when the shells were > 20 mm in size (**Figures 5.3 & 5.6**). Percent loss was 8 % (3 ind) on the 30th day but was increased to 30 % (10 ind) on the 90th day (**Figure 5.7**). Seven of the nine cages were recovered after a typhoon, but needing major repair. Recovery rates did not significantly vary among treatments ($F_{(2, 6)}$ = 0.125, P = 0.885, **Table 5.2**). There was no incidence of tag loss or broken shells in this set up.

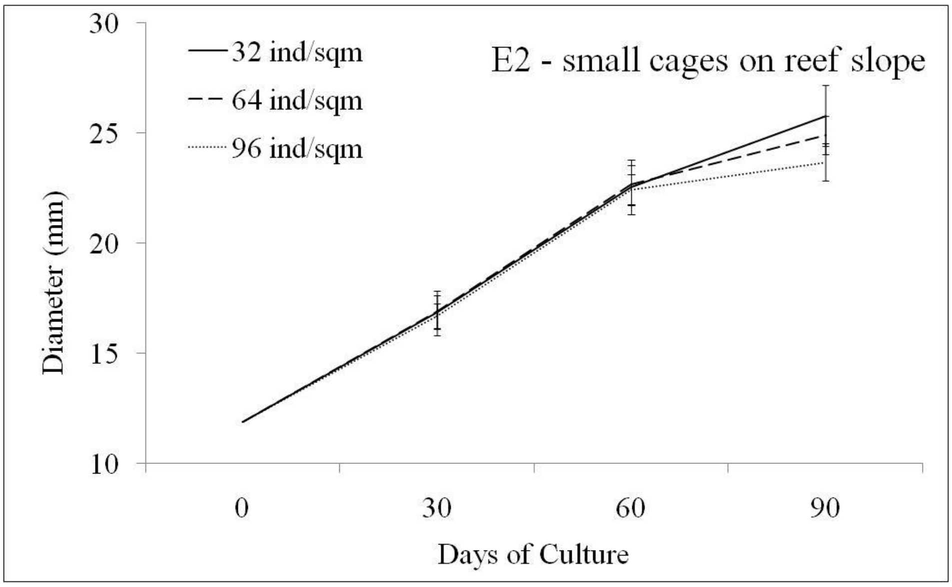

Figure 5.3. Monthly mean diameter (mm) of *T. niloticus* raised at different stocking densities in small cages set on the reef slope (E2). The error bars are 95 % confidence intervals. The initial size at zero day of culture is the average size of all trochus used in the experiment.

Experiment 3 – Semi-indoor tanks

Mean sizes on the 30th day for T2 and T4 were significantly higher than in T1 and T3 in spite that T4 had the highest density ($F_{(3, 988)}$ = 47.12, P = < 0.001). On the 60th, only T2 was significantly higher than the rest of the treatments, despite that the density here was twice higher than in T1 ($F_{(3, 966)}$ = 34.02, P = < 0.001, **Table 5.2**). The mean growth rate in T1 on the 60th day was 2.6 times higher than in T4. Growth rates generally declined with increasing size and density (**Figures 5. 4 & 5.6**). Growth rates among treatments on the 30th and 60th day ranged between 1.8 – 3.9 mm mo^{-1} and 0.8 – 2.1 mm mo^{-1} respectively. Recovery rates were high for all treatments and were not significantly different ($F_{(3, 4)}$ = 0.919; P = 0.508, **Table 5.2, Figure 5.7**). There was no incidence of tag loss in this set up.

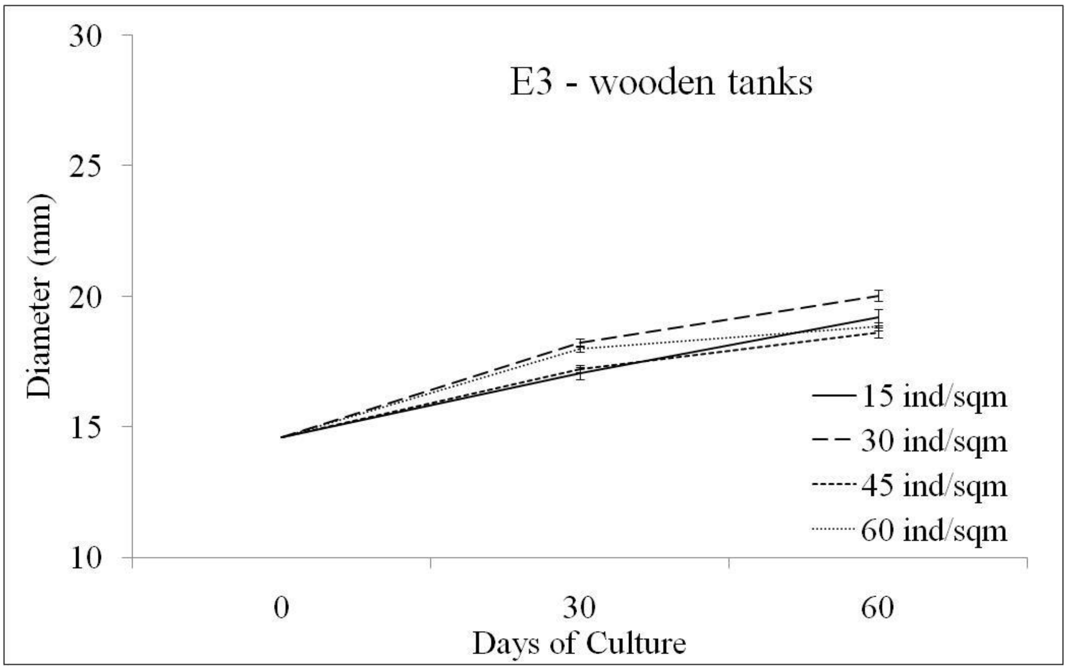

Figure 5.4. Monthly mean diameter (mm) of *T. niloticus* raised at different stocking densities in semi indoor tanks (E3). The error bars are 95 % confidence intervals. The initial size at zero day of culture is the average size of all trochus used in the experiment.

Experiment 4 – Large steel cages on a reef slope

At the end of the culture period, analysis of variance revealed significant difference among treatments in the sizes of trochus ($F_{(3, 671)}$ = 165.92, P = < 0.001, **Table 5.2**), with those in T1 have a mean size of at least 16 % higher than in T4. Growth rates declined with increasing size and density. On the 30th day, growth rates ranged between 1.1 – 2.2 mm mo^{-1} and 0.7 – 1.8 mm mo^{-1} on the 90th day (**Figures 5. 5 & 5.6**). Average recovery rates ranged between 74 – 98 % and were not significantly different among treatments ($F_{(3, 8)}$ = 1.246, P = 0.356, **Table 5.2**).

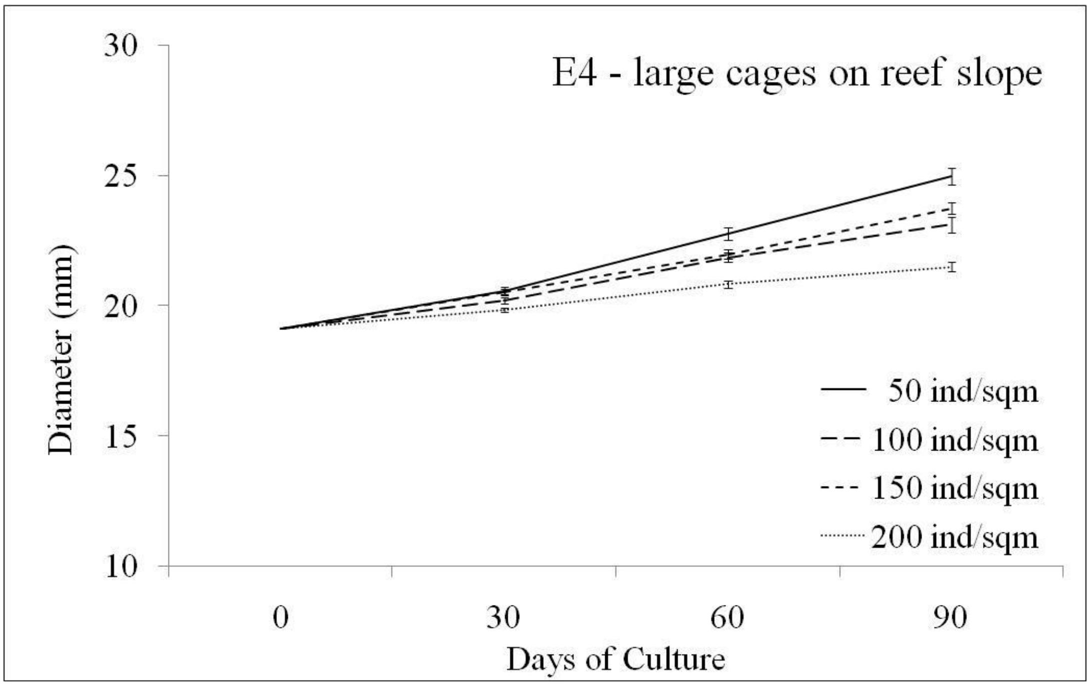

Figure 5.5. Monthly mean diameter (mm) of *T. niloticus* raised at different stocking densities in large cages set on the reef slope (E4). The error bars are 95 % confidence intervals. The initial size at zero day of culture is the average size of all trochus used in the experiment.

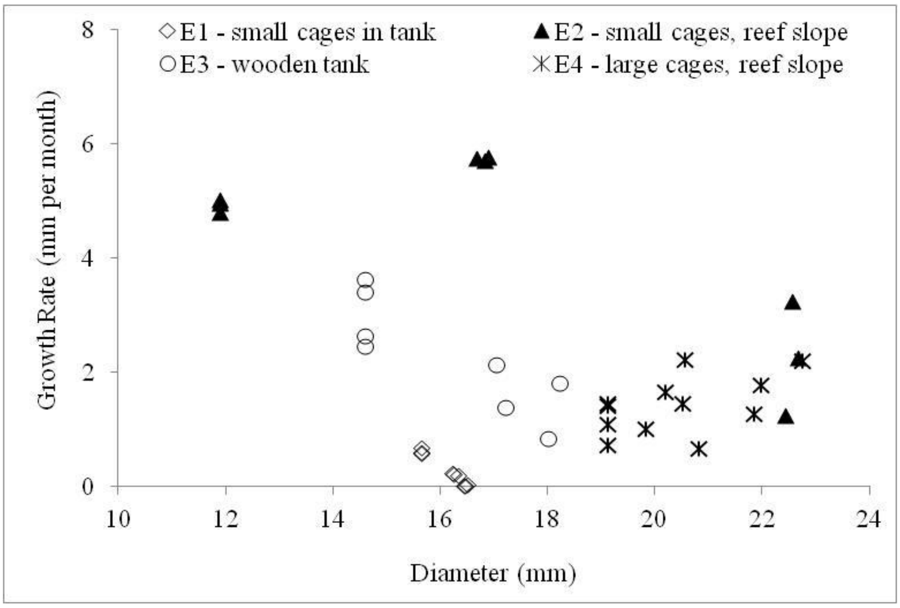

Figure 5.6. Monthly growth rates (mm mo^{-1}) of *T. niloticus* at different stocking densities raised in indoor tanks (E1 & E3) and in cages set on the reef slope (E2 & E4). The diameter of the shell (mm) on the x-axis is an average between the previous and succeeding sampling events.

Percent loss (dead + escaped) was < 10 % on the 60^{th} day due to escape via a small hole on the edge of the net cover, and then it rose up to 30 % on the 90^{th} day because some of the shells escaped through the cages' open window when three cages were blown towards the shore by strong waves associated with a typhoon (**Figure 5.7**). The day before the last sampling, one of the cages was blown and recovered on the shore with only two shells left, but the cage was intact and undamaged, except that the cage's window was opened. Another two cages were turned upside down and had their windows opened allowing some shells to find their way out, however, all cages were undamaged, and none of the recovered individuals had broken shells. Percent loss tag was 1.1 % on the 90^{th} day.

Table 5.2. Mean sizes (± se) and recovery rates (RR) of *T. niloticus* recorded after 30, 60 and 90 or 120 days in different stocking densities or treatments (T) per growth experiment. Within each experiment, stocking densities with the same superscript did not differ significantly.

Experiment Code (no. of replicates)	Duration (days)	T	Mean Diameter (mm) ± se			Mean RR (%)
			30th day	60th day	90th or 120th day	
E1 (3)	120	1			16.54 ± 0.16^{a}	100.00 ± 0.00^{a}
		2			16.41 ± 0.12^{a}	95.83 ± 2.40^{a}
		3			16.46 ± 0.10^{a}	97.22 ± 2.78^{a}
E2 (3)	90	1	16.84 ± 0.50^{a}	22.56 ± 0.60^{a}	25.79 ± 0.66^{a}	88.89 ± 19.24^{a}
		2	16.90 ± 0.36^{a}	22.66 ± 0.44^{a}	24.92 ± 0.42^{ab}	83.33 ± 0.00^{a}
		3	16.69 ± 0.28^{a}	22.44 ± 0.34^{a}	23.68 ± 0.40^{b}	83.33 ± 19.24^{a}
E3 (2)	60	1	17.06 ± 0.12^{b}	19.20 ± 0.16^{b}		99.00 ± 0.00^{a}
		2	18.22 ± 0.08^{a}	20.04 ± 0.11^{a}		98.50 ± 1.00^{a}
		3	17.23 ± 0.07^{b}	18.62 ± 0.10^{b}		96.17 ± 2.84^{a}
		4	18.01 ± 0.06^{a}	18.84 ± 0.08^{b}		99.38 ± 0.12^{a}
E4 (2)	90	1	20.56 ± 0.08^{a}	22.77 ± 0.13^{a}	24.97 ± 0.16^{a}	98.00 ± 2.00^{a}
		2	20.20 ± 0.06^{b}	21.85 ± 0.10^{b}	23.10 ± 0.15^{c}	80.83 ± 15.69^{a}
		3	20.53 ± 0.05^{a}	21.98 ± 0.08^{b}	23.74 ± 0.11^{b}	74.00 ± 7.78^{a}
		4	19.84 ± 0.04^{c}	20.83 ± 0.07^{c}	21.50 ± 0.09^{d}	90.42 ± 6.84^{a}

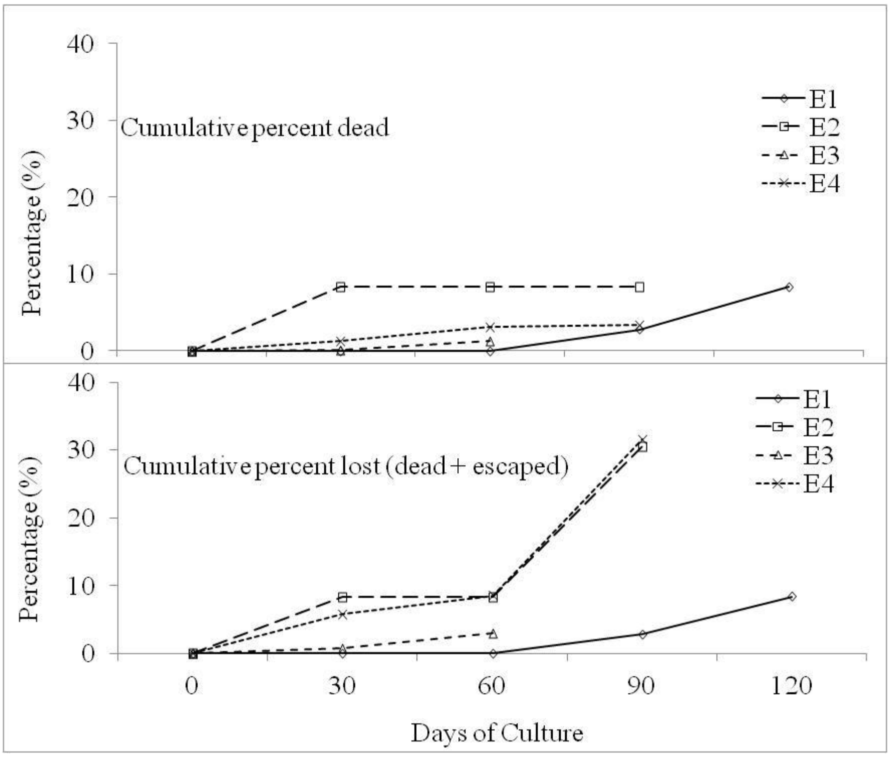

Figure 5.7. Cumulative percent dead measured as recovered empty shells every monthly sampling (top) and cumulative monthly percentage of lost (dead + escaped) *T. niloticus* in four growth experiments conducted in indoor tanks (E1 & E3) and in cages set on the reef slope (E2 & E4).

DISCUSSION

The overall mean (± se) growth rates (0.21 ± 0.02 mm mo^{-1}) of trochus in E1 was 20 times lower than in E2 (4.4 ± 0.19 mm mo^{-1}). The lowest growth rate in E1 suggests that trochus juveniles in that set up did not feed on coconut leaves. Prior to initial stocking at E1, the cages had been in the water for about a month and algae could have grown on the substrate which resulted in a slight increase in size of trochus on the 30^{th} day. Subsequently, newly collected leaves were introduced monthly containing no algae for the trochus to graze on. Water temperature (26 – 31°C) and salinity (30 – 35 ppt) were within the shell's optimum requirement (Gimin and Lee 1997a) and are therefore unlikely to have reduced growth. Experiment 1 and 3 (E1 and E3) were conducted indoor and the outdoor experiments (E2 and E4) were conducted at the same months and duration and months thus we disregard the possible variation of influence of environmental parameters on the growth and survival of trochus at each experiment.

The high growth rates in E2 as compared to other growth experiments (E1, E3 and E4) could be due to the abundance of detritus that settled on coconut leaflets used as substrate. Trochus in their natural habitat often feed on encrusting and filamentous algae and detritus (Heslinga 1981; Nash 1993; Lambrinidis et al. 1997b; Clarke et al. 2003), so that it is presumed that trochus fed on settling organic solids instead of algae which may not immediately grow on coconut leaflets. Feeding marks on organic materials that settled on the substrate were evident during sampling as also noted in E4 (**Figure 5.1 H**). The high growth rate of trochus in the first 60 days (> 5 mm mo^{-1}) was comparable to the growth of juveniles in the wild (Smith 1987; Hoffschir 1990; **Chapter 6**) and under laboratory condition (Heslinga 1981; Hoang et al. 2008) with adequate feeding.

The sudden drop in the average (± se) growth rate of trochus in all treatments in E2 from 5.7 ± 0.14 mm mo^{-1} on the 60^{th} day to 1.9 ± 0.17 mm mo^{-1} on the 90^{th} day is presumably due to food shortage. The amount of substrate was maintained throughout the culture period, but the demand for food had increased with shell size. As a possible effect of food shortage, the decline in growth rate in the treatment with the highest density (T3) is clearly noticeable in **Figure 5.3** and the mean size of trochus in T3 was significantly smaller than in T1 (**Table 5.2**). In T3 the average (± se) growth rate declined sharply from 5.7 ± 0.25 mm mo^{-1} on the 60^{th} day to 1.0 ± 0.16 mm mo^{-1} on the 90^{th} day **(Figure 5.6).**

In E3, growth was generally not affected by densities in that there were no significant differences among T1, T3 and T4 after 60 days of rearing. The highest average growth rate (3.6 mm mo^{-1}) obtained here for all treatments were still slower than other growth studies (~5 mm mo^{-1}) (Heslinga 1981; Hoang et al. 2008) suggesting that the food required for optimum growth had not been available. By visually comparing the growth rates in E2 and E3 (**Figures 5.3, 5.4 & 5.6**), the latter had lower growth rates despite treatments in E3 having lower densities than in E2. Most of the algae on corrugated plates were overgrazed on the 30th day and additional food came only through the introduction of pre-soaked coconut leaves which presumably did not meet the shell's increasing demand for food.

Trochus in E4 did not grow as fast as in E2 although both experiments were deployed at the same depth and time. If we examine **Figure 5.5**, differences in growth between treatments in E4 gradually increase over time. A similar trend was also noted in E2 at the end of the study when the animals were > 20 m in size (**Figure 5.3**). Looking into the growth rate at certain size (**Figure 5.5**), trochus in E4 and E2 at > 20 mm in size had relatively similar growth rates. Growth of trochus of such size in the wild could be at an average of 5 mm. mo^{-1} (**Chapter 6**). It is then possible that the slow growth was due to food shortage because the densities in E2 ranged between 32 and 96 ind m^{-2} and were comparable to the densities of 50 and 100 ind m^{-2} in T1 and T2 in E4.

With adequate food supply, trochus under laboratory conditions can grow as fast as in the wild or even faster than in sea cages. Trochus provided with algae and detritus attached on coral rocks reached 62 mm in 12 months (Heslinga 1981). Hoang et al. (2008) obtained similar growth rates in rearing 12 mm juveniles fed with algae growing on rocks for 3 months, and larger individuals (mean size: 28 mm) fed with a combination of dried algae, soya bean, detritus and *Nitzschia* sp. for a period of 2.5 months. Rearing in tanks however does not produce high survival rates (Etaix-Bonnin and Fao 1997; Gonzales et al. 2009). The increasing variability in size of trochus after an extended culture period in tanks (Heslinga 1981) suggests that larger individuals tend to grow faster while the small ones remained stunted with time. The success of large over small individuals in competing for food has been found among hermit crabs (Ramsay et al. 1997). However it is not always the case that the adults win in the intraspecific competition. Among the intertidal gastropod *Nerita atramentosa*, increased densities due to high recruitment have caused a reduced in growth of juveniles but have increased mortality among adults (Underwoon 1976). Among the limpet *Cellana tramoserica,* the growth and survivorship of small limpet decreased abruptly when caged with other small individuals, but were not affected by competition with large individuals. The radular teeth of juveniles were shorter and narrowly spaced than those of large limpet, making them to efficiently feed on algae growing on highly pitted surface of their rock habitat (Marshall and Keough 1994). Competition for food could be one of the reasons for high mortalities, but the high survival rates of trochus in E1 despite of undetected growth for 160 days suggests that factors other than food is causing such mortalities. Huchette et al. (2003) found a negative correlation between levels of ammonia and abalone growth in tanks, and they further explained that ammonia can cause stress and affect the health of the cultured shells.

Although we obtained a high survival rates in indoor tanks, rearing large numbers of trochus in land based tanks is more costly (Heslinga 1981) than in sea cages (Amos and Purcell 2003). If trochus are raised in cages after reaching a size of 5 mm, then with a conservative growth rate of 4 mm mo^{-1}, it would only take half a year for the shells to reach a size (> 30 mm) suitable for release. This is three or four times faster than when rearing the shells in indoor tanks as using the standard methods of the WPU-SF (Gonzales et al. 2009).

The high recovery rates in our reef cages compared to other trochus cage culture could be related to the substrate used, the design of the cages and depth at which our cages were installed. The cages of Amos and Purcell (2003) used coral rocks and were at 0 to 2.5 m deep during spring tides which could have been impacted with big waves. When agitated, coral rocks used as substrate in cages can cause damage on trochus and on the cages. Our cages were only 5 –6 m deep on the reef slope, so some units were blown to the shore when harsh weather came. However, the large cages were still intact and the recoverd shells were in perfect condition because the leaves substrate did not cause any damage to the shells or the cages. Provision of anchors could help keep the cages in place and avoid the loss of stock during bad weather.

Implications for management

Several authors (Amos and Purcell 2003; Clarke et al. 2003) have suggested an ideal trochus stocking density of 30 ind m^{-2} in sea or reef cages, but our results in E2 in the first 60 days agree with Hoang et al. (2007) that 10 – 22 mm juveniles can be raised in cages at 100 ind m^{-2} with a reasonably high growth rate of around 5 mm mo^{-1}. Most intermediate cage culture studies used coral rocks as substrate (Purcell 2001; Amos and Purcell 2003; Hoang et al. 2007) for which periodic changing could be difficult in times of bad weather and when availability of algae covered rocks is limited (Hoang et al. 2007). Therefore, using coconut leaves as substrate could be more advantageous as they are light, easy to install or remove from the cage and reduce the risk of shells being smashed by the rocks during cage deployment or sampling. Cages with plant substrate are light, hence easy to transport to desired places.

The need to reduce stocking densities as trochus increases in size (Hoang et al. 2007) means additional cages, costs and labour. This problem could be remedied by increasing the amount of coconut leaves as substrate which would offer more surfaces for settling organic particles, or by introducing of filamentous algae for the trochus to feed on.

Food requirement of trochus may change with size or age. Adult trochus feed on a variety of plants and animals (Rao 1937; Lambrinidis et al. 1997a) and may exibit food preferences in the wild (Isa et al. 1997). Therefore, the introduction of soft filamentous seaweeds in cages as culture period progresses or as the animals grow bigger would help correct the declining growth rate associated with the increasing size of the shell and length of culture period.

Purcell (2001) have successfully increased growth of trochus by adding more food.

Trochus in the wild with sizes 30 – 40 mm can grow at an average of 5 mm mo^{-1} (Smith 1987; **Chapter 6**), suggesting that the declining growth of trochus in cages as culture period progresses is a result of food shortage. Although we used > 10 mm juveniles in our study, in Japan, smaller in size (4 mm) juveniles raised in intermediate basket set in a seawater pond for 61 – 71 days had a growth increment of 1 – 2 mm mo^{-1}. Trochus raised in basket hung in the sea grew from 4.2 mm – 22 mm (2.4 mm mo^{-1}) in 211 days (Isa et al. 1997). Assuming such growth rates, it would take 10 months of rearing to make the shell reach a suitable release size of 30 mm. It is not clear whether these authors used feeding or artificial food in this study. At such an age, trochus feed on diatoms which may settle or grow on the net. Without any substrate however, the amount of settling diatoms could be limited and may not satisfy the food requirements of the stock.

With the growth rate obtained in small cages in the current study, intermediate rearing of 5 mm juveniles in subtidal cages with coconut leaves as substrate would only take 5 – 6 months for the shell to reach a suitable release size of 30 mm.

Trochus raised with coconut leaves as substrate developed a reddish streak on their shells (especially in E3; **Figure 5.1E**) that is not found in their wild counterparts. Lindberg and Pearse (1990) provided evidence that the colour of limpet shells is influenced by their food, which suggests that parts of the coconut leaves were ingested by the trochus while feeding on algae or detritus thus giving the shells its unique colouration. Coconut leaves therefore can be an alternative means of diet tagging instead of using formulated feeds used in Gallardo et al. (2003) which when excessively used can have harmful effect on the environment (Chua Thia et al. 1989; Páez-Osuna et al. 1998). When this marking remains with age, this could be useful in identifying hatchery produced trochus released in the wild.

The success of trochus reseeding depends upon the number of individuals that reach maturity and are able to breed (Nash 1993), so careful consideration should be undertaken in trochus restocking. While intermediate rearing was successfully carried out on the deep reef slope, we still recommend to release the shells at the intertidal area, considering the high mortality rate for sub-adult trochus at sub-tidal reefs (Villanueva et al. 2010). As we have found that trochus growth in reef cages could be as fast as those at the intertidal area, it is likely that larvae that settle in the subtidal environment have less chances of surviving due to predation compared to those that settle in intertidal habitats (Nash 1993). Acclimation of hatchery-produced trochus to potential predators in the wild could help reduce mortalities. Exposing intertidal whelk to a known predator effluent has caused the shells to increase in thickness and decrease somatic growth (Edgell and Neufeld 2008). Also conditioning of hatchery-produced crabs to fish predators has increased their survivorship in the wild (Davis et al. 2005). Shelter acclimations have also increased the survivorship of hatchery-produced fish upon releasing in the wild (Kawabata et al. 2011). It is also worth mentioning that the release of hatchery produced juveniles is only recommended in the absence of wild trochus for restocking (Bell et al. 2005) and in conjunction with careful management of harvesting.

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Chapter 6

TRANSLOCATION OF WILD *Trochus niloticus*: PROSPECTS FOR ENHANCING DEPLETED PHILIPPINE REEFS

Roger G. Dolorosa, Alastair Grant and Jennifer A. Gill

ABSTRACT

The large gastropod *Trochus niloticus* of coral reefs in the Indo-West Pacific Region is widely harvested for its meat and shell. They have a patchy distribution, but decades of translocation to many tropical oceanic islands in the Pacific have expanded its limited distribution range, and it has become an important resource in some of those areas. Trochus is highly susceptible to over-harvesting and it is now threatened in the Philippines where it is naturally occurring. Although harvesting of trochus is now prohibited, uncontrolled exploitation continues, further threatening the remaining population with the risk of local extinction. Trochus have short larval period and limited mobility after settlement so that population recovery in overexploited offshore reef areas could be impossible even when it is spared from over-harvesting. Conserving of trochus through the release of hatchery-produced juveniles may not be feasible because of high cost and low survival rates, but the translocation of wild trochus appears promising, with high survival rates and varied growth rates in some sites. Rebuilding wild trochus population through translocation into a network of well-managed marine reserves may be the best potential option for reviving the country's depleted trochus populations. As trochus graze on algae, their revival can also help to protect coral reefs from algal overgrowth, and thus further enhance the enormous goods and services derived from reef ecosystems. Success in reviving trochus populations by translocation may also pave the way to similar conservation strategies for other important reef invertebrates.

Keywords: growth, marine protected areas, reintroduction, survival

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INTRODUCTION

The once abundant reef gastropod *Trochus niloticus* (or trochus as it is commonly called) populations in the Philippines have been over-harvested (Hahn 2000; Gapasin et al. 2002; Floren 2003) to the extent that their exploitation has been prohibited through national legislation (DA 2001). In spite of legislation requiring the protection and conservation of the species, on-going trochus conservation by releasing hatchery-produced juveniles is only undertaken in the province of Palawan (Gonzales et al. 2009; Avillanosa et al. 2010), where the remaining natural populations are more abundant than in other areas of the country (Schoppe et al. 1998; Del Norte-Campos et al. 2003; Dolorosa et al. 2010).

The practice of releasing hatchery-produced juveniles as a means to revive a depleted trochus population is not new. Many trochus-producing countries have engaged in the artificial production and release of trochus juveniles but with limited success (Chauvet et al. 1997; Isa et al. 1997; Lee 1997; Lee and Toelihere 1997; Purcell et al. 2004; Purcell and Cheng 2010). Hatchery propagation projects require a large amount of funds to build and operate (Lee and Lynch 1997) and juveniles suffer high mortalities upon release in the wild (Castell 1993; Isa et al. 1997; Crowe et al. 2002). This requires the intermediate culture of juveniles in sea cages until a size suitable for release is reached, but this method is still beset with problems like low survival and poor cage design (Purcell 2001; Amos and Purcell 2003; Hoang et al. 2007).

The restocking of hatchery-produced juveniles in the Philippines has been employed as a conservation measure, but these approaches were either short-lived or had unsuccessful outcomes. Trochus breeding for restocking in Palawan, Philippines was first started by a Japanese corporation - Iris Marine Development Corporation (IMDC) between 1999 and 2008. However, in spite of success in breeding and releasing of millions of larvae and juveniles, there was no harvest of the released trochus even in the well-guarded reef in front of the hatchery

(**Chapter 3**), possibly because of high mortality of released juveniles (generally 3 – 30 mm diameter) and uncontrolled gleaning of local residents in other sites of release. In 2000, the Southeast Asian Fisheries Development Centre-Aquaculture Department (SEAFDEC/AQD) had also engaged in trochus breeding using some parent shells from IMDC, as a result, thousands of juveniles were produced and diet tagged, of which 3 000 individuals were transported back to Palawan and released in a village sanctuary (presumably Sabang Reef) in Binduyan (Gapasin et al. 2002; Gallardo 2003), a northern village of Puerto Princesa City. The SEAFDEC/AQD trochus breeding program however was cut short in 2003 because of limited funds and species prioritisation (Gapasin pers. comm.).

When IMDC closed down in 2008, the hatchery facilities were donated to the Western Philippines University (WPU) and is now called WPU-Samong Farm (WPU-SF). The university continues the hatchery operations through a fund coming from the Department of Science and Technology – Philippine Council for Aquatic and Marine Research and Development (DOST-PCAMRD), and releases a total of 2 000 4–month old juveniles (about 8 – 15 mm diameter) in 2009 in a peoples' sanctuary called Sabang Reef in Binduyan, Puerto Princesa City. However, our visit to this site six months after the release failed to find any survivors. This is not surprising because the survival of hatchery-produced juveniles (size range: 5 – 25 mm) upon releasing in the wild could be very low (Castell 1993; Chauvet et al. 1997) or even zero survival for 8 – 16 mm juveniles 4 months after the release (Isa et al. 1997).

The complexity and high cost in hatchery propagation and intermediate rearing of hatchery-produced trochus in sea cages combined with uncertain survival in the wild mean that a cheap and effective method of restocking is needed to help revive trochus populations, especially in a developing country like the Philippines in which it is an important source of revenue. In the early 1990s, the government of Palau had shifted already from the costly mariculture and reseeding approach in favour of more cost-effective methods of managing their trochus resources (Amos 1997).

The translocation, reintroduction or relocation of organisms has been widely practiced for the purpose of conserving a threatened species or for increasing the harvest of commercially important species in the wild (Hodder and Bullock 1997 and references therein). As for trochus, more than six decades of translocations of wild populations have been successful in some offshore reefs in the South Pacific where they don't occur naturally (Sims 1984; Smith 1987; Gillett 1993; Ponia et al. 1997; Zoutendyk 1997). This suggests that the revival of heavily exploited trochus populations could be attained by reintroduction of wild captured trochus from other places or the aggregation of trochus within such reef, combined with the effective protection of trochus at the site of release. In this study, we explore the potential effectiveness of translocating wild trochus to enhance a depleted reef. Specifically, we investigated the growth and survival of: a) wild juveniles and adult trochus translocated to a heavily exploited reef; b) non-translocated adult trochus occurring in high density at an unexploited reef; and c) adult trochus translocated to a protected site with few occurring individuals.

METHODS

The study sites

The study was carried out in three sites in Palawan, Philippines. Site 1 was in a coastal reef (10° 01.379' N and 119° 6.337' E) near the WPU-SF hatchery in Binduyan, one of the northern most villages of Puerto Princesa, the capital city of the province of Palawan. The reef slope in Site 1 was narrow, extending no more than 100 m from the shoreline which slopes gently to a scale of 8 – 10 m deep and then vertically at the edge, forming a cliff-like wall. The lower intertidal area was thickly covered with a mixture of large metamorphic and sedimentary rocks. The subtidal area was generally characterised by coral rocks with an increasing proportion of live corals towards the edge of the slope (**Figures 6.1 and 6.2 A & B**). Commonly encountered reef invertebrates include octopus and some species of shelled gastropods. This site is likely to be a favourable habitat for trochus as it is directly exposed to high energy waves (Nash 1993) brought about by the monsoons.

Being adjacent to the hatchery, Site 1 served as the release area for about 6 million larvae and 0.5 million juveniles (3 – 30 mm length) produced by the IMDC before the hatchery was donated to WPU. However, survey of the subtidal area did not reveal the presence of any large trochus, although local residents that were gleaning along the shore in front of the IMDC hatchery occasionally found 20 – 30 mm juveniles which they handed in to the hatchery staff (**Chapter 3**).

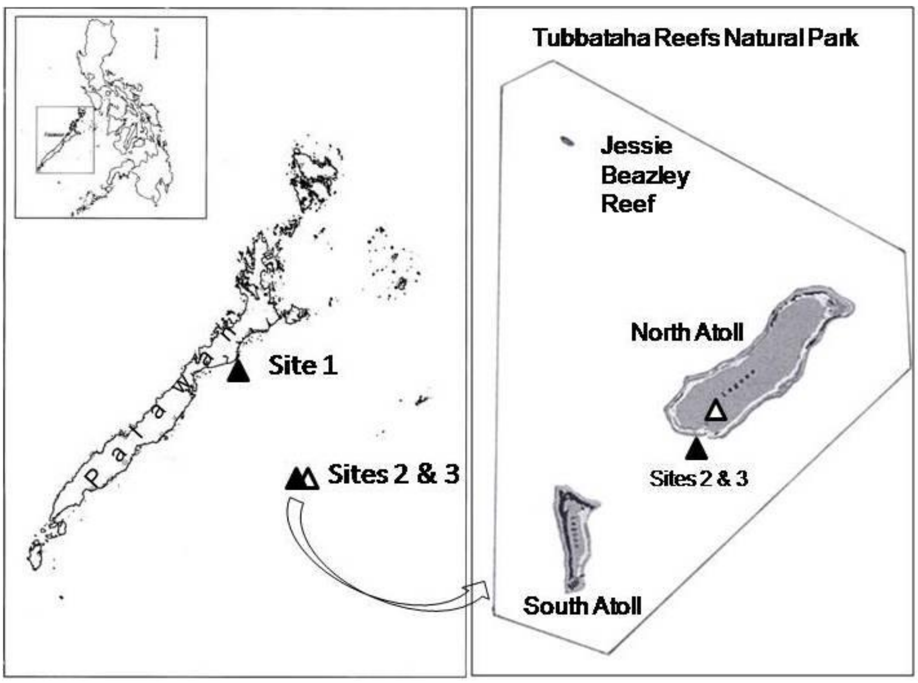

Figure 6.1. Maps showing the three study sites in Palawan. Site 1 in Binduyan (black triangle), Site 2 (black triangle) and Site 3 (white triangle) at TRNP.

Sites 2 and 3 were at the North Atoll of the Tubbataha Reefs Natural Park (TRNP). TRNP is a marine park located at the middle of the Sulu Sea. The park covers nearly 100 000 ha and approximately 10 % of the area is covered by coral reefs (TMO 2011). The climate in the park is influenced by the monsoons, characterised by rough seas and heavy rain showers during most months of the year, with a period of calmness only during the months of March – June. Trochus at TRNP occurred abundantly at the seaward reef of the North and South Atolls (Dolorosa and Schoppe 2005; Dolorosa et al. 2010; Jontilla et al. in press) with few individuals found inside the lagoon of these two Atolls (pers. obs.).

Because of illegal fishing in the park (Dolorosa et al. 2010; Jontilla et al. in press), two areas (Site 2 and Site 3) near the Ranger Station (RS) were chosen as study sites to protect the studied trochus against illegal harvesting . Site 2 (8^{o}50.875' N, 119^{o}54.960' E) was at the seaward reef area characterised by a narrow strip (about 50 – 80 m wide) of massive and sub-massive corals. At the shallow subtidal area where most trochus were found, the corals stand more than a meter high and their heads are nearly exposed at extreme low tide (**Figures 6.1 & 6.2 C**). Sizes of coral colonies increase towards the edge of the reef slope. The drop-off extends to a depth of about 1 000 m. This area held one of the highest densities of trochus (Dolorosa et al. 2010; Jontilla et al. in press) and had been a release site for some trochus confiscated from illegal fishers during the years 2007 – 2008 (Conales pers. comm.).

Figure 6.2. The release sites for trochus: Site 1 (A & B), Site 2 (C) and Site 3 (D).

Site 3, located inside the lagoon about a kilometre east from the RS, was no more than 3 m deep at high tide and about 50 m in diameter. The substrate was composed of rubble with a few coral rocks. Surrounding the release site were large colonies of branching (*Acropora* spp.) and massive (*Porites* spp.) corals, and sand patches. The tall coral colonies surrounding the area both served as barriers and hiding places for trochus, while the sand patches at the deeper side of the area may inhibit the trochus from moving towards the deeper part of the lagoon (**Figures 6.1 & 6.2 D**). Most branching corals were dead due to a crown-of-thorns (COT) starfish outbreak from 2008 until the time of the study. The remaining living corals were the large colonies of *Porites* spp. and small table corals *Acropora* spp. thriving on their flattened heads. There were quite a few naturally occurring trochus in Site 3 with two or three young adults (about 80 mm in diameter) noted at the time of translocation of marked trochus. The entire lagoon has not been surveyed but our observation in some sites showed that it is generally composed of sandy substrate with patches of coral colonies along the shallow edges.

Duration of the study and sampling of translocated and non-translocated trochus

Two groups of trochus were released in Sites 1 and 2 and only one group was released in Site 3 (**Table 6.1**). The first group in Site 1 were collected in a heavily gleaned intertidal area of Bitauran Reef about 6 km away from the area of translocation. Collections were conducted once a month when the reef was exposed during day low tide starting October 2009 until July 2011. The collected shells were marked and released within the day of collection along the rocky shoreline in Site 1. A total of 595 individuals were collected over a period of 10 months of which 532 individuals were tagged and released at the intertidal area in front of the IMDC hatchery.

Table 6.1. Duration and sampling schedules at the three study sites in Palawan, Philippines. T – Translocated, NT – non-translocated

	Binduyan Site 1		TRNP, Palawan Site 2		Site 3
	Group 1	**Group 2**	**Group 1**	**Group 2**	**Group 1**
Remarks	T	T	NT	NT	T
No. of sample	532	125	403	83	108
Date started	9 Oct 2009	13 Nov. 09	18 Oct. 09	2 Jan. 10	14 Dec. 09
Date finished	10 Aug. 10	13 Aug. 10	15 June 10	15 June10	16 June 10
Duration (days)	278	270	242	164	184
Sampling	Monthly	Monthly	Dec, Apr, & Jun	Apr, & Jun	Apr, & Jun

The second group in Site 1 were composed of large individuals collected by contracted octopus fishermen from the nearby village. Because of fear of losing the trochus upon releasing during the typhoon season and to allow time to inform the community about the on-going project, these shells were firstly kept in one intertidal tank provided with steel matting cover at a density of 32 ind m^{-2}. After two months in captivity, 125 live individuals (97 %) were still alive. The recovered live shells were tagged, measured for diameter and released at the lower intertidal part of the reef in front of the IMDC hatchery in November 2009 (**Table 6.1**). Releasing was timed with afternoon low tide to avoid possible stress due to excessive heat during the day. All marked trochus were placed in an upright position beside rocks between two intertidal tanks and were allowed to naturally disperse.

In Site 2, a total of 486 individuals were collected at the seaward reef area near the Ranger Station, marked and immediately returned on the same day in the area of collection. Collection and marking were conducted in two batches. The first batch or group composed of 403 individuals were collected in October 2009, and the second batch or group composed of 83 individuals were tagged in January 2010 (**Table 6.1**).

In Site 3, a total of 108 trochus were collected near Site 2 in December 2009. These were marked and sampled before translocating them in Site 3 (**Table 6.1**).

Monitoring of trochus after the release was conducted monthly at Site 1 considering the proximity of the site, while in Sites 2 and 3, monitoring was timed with the Rangers' shift which happened once every two months or during research expedition in summer. In Sites 2 and 3, sampling was conducted every 2 – 4 months. The study in Site 1 started in October 2009 and ended in August 2010 (278 days). For large trochus in Site 1, the study started in November 2009 until August 2010 or for a period of 270 days. In TRNP, there were two groups of trochus in Site 2, the first group was tagged in October 2009 and the second group was tagged and released in January 2010. The sampling of two groups of trochus in Site 2 ended in June 2010 or after a period of 242 and 164 days respectively. In Site 3, the interval between the first release and final sampling was 184 days (**Table 6.1**).

Recapture of juveniles (Group 1, Site 1) was conducted during calm high tide because the area of release was not totally exposed at low tide. About 2 hours of searching for juveniles hiding beside rocks was conducted no more than 50 m (along the shore) from the point of release. Although the juveniles tend to hide under the rocks, none of the rocks were moved (or turned upside down) to avoid habitat disturbance. For Group 2 in Site 1, recapture during calm low tide was done in shallow subtidal reef reachable by breath-hold dive. The 2 hours search extended no more than 100 m from the point of release.

In Sites 2 and 3 recapture was done when the sea was calm especially during low tide. Recapture was done by firstly locating the point of release with the aid of a GPS. After which, snorkelling was conducted for about 1 – 2 hours. Because of abundance of unmarked trochus and thick marine growth on shells in Site 2, any trochus encountered were examined carefully with the presence of tag. At one time, SCUBA diving at the deeper part (~10 m deep) of Site 2 was conducted to examine the presence of marked shells. Recapture of trochus by snorkelling in Site 3 was only about an hour because the area was dominated by rubble and sparse dead corals. The shells were not covered with marine growth and there were very few other shells aside from the marked ones, thus searching and recapture at Site 3 was not as complicated as in Site 2.

Measurement of trochus during the initial sampling and subsequent sampling events were conducted in a shelter to avoid stress due to sun exposure. The maximum basal diameter of each shell was measured with sliding callipers to the nearest 0.1 mm. Trochus with damaged but readable tag numbers at recapture received an additional tag. A pencil marking on the aperture of each shell was re-inscribed at every recapture although, at Sites 2 and 3, the pencil markings were found to last more than 3 months. No measurement was taken from dead individuals as the time of death was not known. Recapture and release was made on the same day to reduce stress.

The release of the shells was usually done during afternoon low tide when the weather was cool. In Site 1, the trochus were released in one specific point beside an intertidal concrete tank, while in Sites 2 and 3 the trochus were randomly placed beside rock habitats by snorkelling. During the release, each shell was placed in an upright position, allowing them to immediately hold on the rock substrate without being dislodged by wave or water currents.

Tagging of trochus before translocation or returning in their natural habitat

The tagging procedure for large trochus followed that of Nash (1985) and Lemouellic and Chauvet (2008), with some modifications as follows:

1. Scrape off with a chisel any marine growth and a layer of periostracum on a 5 x 5 cm portion of the shell.
2. Dry the shell with cheese cloth; allow further drying for about 15 – 30 minutes while keeping them in an upright position in plastic trays lined with a damp cloth. Do not expose the shells to sunlight.

3. Apply a small amount of marine epoxy on the cleaned and dried shell's surface; lay a 3 x 1.2 cm TZ extra strong adhesive yellow tape printed with a desired number or code using Brother P-Touch PT1260 printer. For juveniles and sub-adults, a 6 mm wide TZ extra strong tape was used. The length of the tag varied between 25 – 50 mm depending on the width and printed numbers or codes.
4. Apply additional marine epoxy to cover the sides of the tag leaving only the marked surface.
5. Keep the shells in plastic trays lined with a moist cloth for about an hour or until the resin has hardened. Do not expose the shells to direct sunlight.
6. In addition to the external tag, large shells received a pencil inscribed number on their aperture in order to easily identify the shell in cases where its TZ tag is damaged or lost.

For juveniles, the tagging procedure was similar to that used with larger ones, only that there was no scraping of shells, no use of marine epoxy and no pencil marking. A detailed tagging procedure for juveniles is explained in **Chapter 5**.

Data analyses

To determine the growth parameters of trochus, we firstly considered all the sampled sizes at each site. Because we failed to include juveniles and sub-adults in Sites 2 and 3, we grouped all large individuals at each site based on the smallest recaptured size of 68 mm in Site 3 and compute the growth parameters at each site separately. The growth parameters (K and $L\infty$) for each group per site was obtained with the formula of Fabens (Ogle 2010):

$$Lr = Lm + (L\infty - Lm)(1 - e^{-k\delta t})$$

where Lm is size at the time of marking; Lr is size at the time of recapture; δt is the difference between the time at recapture and time at marking or between the second and first recaptures. The time interval (days) were standardised into years by dividing it with 365 days per year. The other parameters are defined under the von Bertalanffy growth formula (Ogle 2010). To determine the size of trochus at each particular age, the estimated growth parameters (K and $L\infty$) were then substituted into the von Bertalanffy growth model:

$$Lt = L\infty \ (1 - e^{-k(t-to)})$$

where Lt is the size at time t; $L\infty$ is the average maximum attainable basal diameter of the shell in mm; K is the Brody growth coefficient, and t_0 is the theoretical age at size zero. The method of Fabens had been revised to include parameters that represent the mean growth rate at arbitrary ages (see Ogle 2010), but because of the within-site variability in growth of trochus we did not apply this revision. The computation was carried out using the FSA package in R version 2.12.0 (Ogle 2010). The large number of cases in which individuals growth increments were zero precluded the use of linear regression to determine the $L\infty$ and K values.

Estimating survival and recapture probabilities of translocated wild trochus in Site 1 were conducted separately for group 1 and 2. At Site 2, a separate analysis was also conducted for each group because of the differences in time intervals between the initial marking and recapture. In determining the survival and recapture probabilities, we used the Cormack-Jolly-Seber Model within Program Mark (White and Burnham 1999; Cooch and White 2010). Four candidate models in which annual survival (ϕ) and recapture (p) probabilities were either constant (.) or time (t) dependent were explored: (1) ϕtpt – time dependent survival and recapture probabilities; (2) ϕtp. – time dependent survival probabilities with constant recapture probabilities; (3) ϕ.pt – constant survival probabilities and time dependent recapture probabilities; and (4) ϕ.p. – constant survival and recapture probabilities. Intervals of recapture in days were divided with 365 days to standardise the recapture interval in year. Each model was examined for Akaike information criterion (AICc), AICc weights and Model Likelihood. In cases where one model with the best fit was chosen, goodness of fit was determined by bootstrapping to determine the probability of finding the deviance in that model. Model averaging was undertaken when at least two models fitted the data well as indicated by each model's AICc weight or the models with lowest AIC value. The different models with their corresponding AIC, model likelihood, deviance and goodness of fit which we used to determine the survival and recapture probabilities for each group of trochus are presented in **Table 6.2**.

Table 6.2. Types of models used to determine the survival and recapture probabilities of translocated and non-translocated trochus.

Site	Group	Model	AICc	AICc Weight	Model Likelihood	Deviance	Goodness of Fit
1	1	ϕtpt	1535.86	1.00	1.00	330.41	$P < 0.001$
	2	ϕ.pt	800.27	1.00	1.00	110.28	$P < 0.05$
		ϕ.p.	930.90	0.56	1.00	0.12	
2	1	ϕ.pt	932.80	0.22	0.39		
		ϕtp.	932.80	0.22	0.39		
	2	ϕ.pt	220.19	0.57	1.00		
		ϕ.p.	220.74	0.43	0.76	2.64	
		ϕ.p.	276.97	0.55	1.00	0.29	
3	1	ϕ.pt	278.85	0.23	0.41		
		ϕtp.	278.75	0.23	0.41		

(ϕ) – Survival probability; (p) – recapture probability; (t) – time dependent; (.) – constant over time.

RESULTS

Size structure of translocated and non-translocated trochus

The first group of trochus in Site 1 was composed of juveniles (< 50 mm) and small adults (50 – 70 mm in diameter). A total of 595 individuals were collected over a period of 10 months of which 532 individuals with an average (± sd) diameter of 34.5 ± 8.2 mm were tagged and released. Of the 532 individuals, 31 (6 %) were small adults with a mean (± sd) size of 55.6 mm (± 5.4). The average size ± sd (90.6 ± 10.9 mm) of Group 2 in Site 1 was nearly three times larger than those in Group 1 (**Table 6.3, Figures 6.3 & 6.4**).

Table 6.3. Sizes of translocated (Sites 1 and 3) and non-translocated (Site 2) trochus from Palawan, Philippines. T – translocated; NT – non-translocated.

Parameters	Binduyan Site 1		TRNP, Palawan Site 2		Site 3
	Group 1 T	Group 2 T	Group 1 NT	Group 2 NT	Group 1 T
No. of samples	532	125	403	83	108
Mean size (mm)	34.3	90.6	93.5	82.0	99.9
sd	8.4	10.9	11.0	18.7	10.9
Min. size	12.2	57.3	61.3	47.6	64.0
Max. size	76.1	121.5	129.0	121.3	128.0
95 % CI	0.7	1.9	1.1	4.1	2.1

In Site 2, the average (± sd) diameter (93.5 ± 11.0 mm) of Group 1 was a little larger compared to Group 2 (82.0 ± 18.7 mm). In Site 3, the marked trochus had a mean (± sd) size of 99.9 ± 10.9 (**Table 6.3, Figure 6.3**).

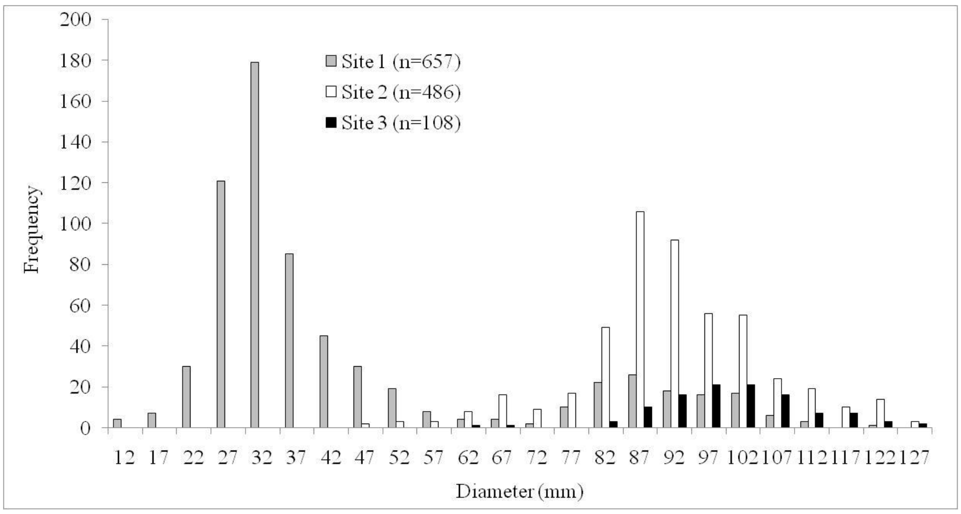

Figure 6.3. Variation in size structure of translocated (Sites 1 and 3), and non-translocated trochus (Site 2). The number of samples in Site 1 (n = 657) is the sum of samples from group 1 (n = 532) and group 2 (n = 125). The high number of small-sized (< 50 mm in diameter) in Site 1 was a result of monthly collection, marking and releasing, while it was difficult to find small-sized trochus in Sites 2 and 3.

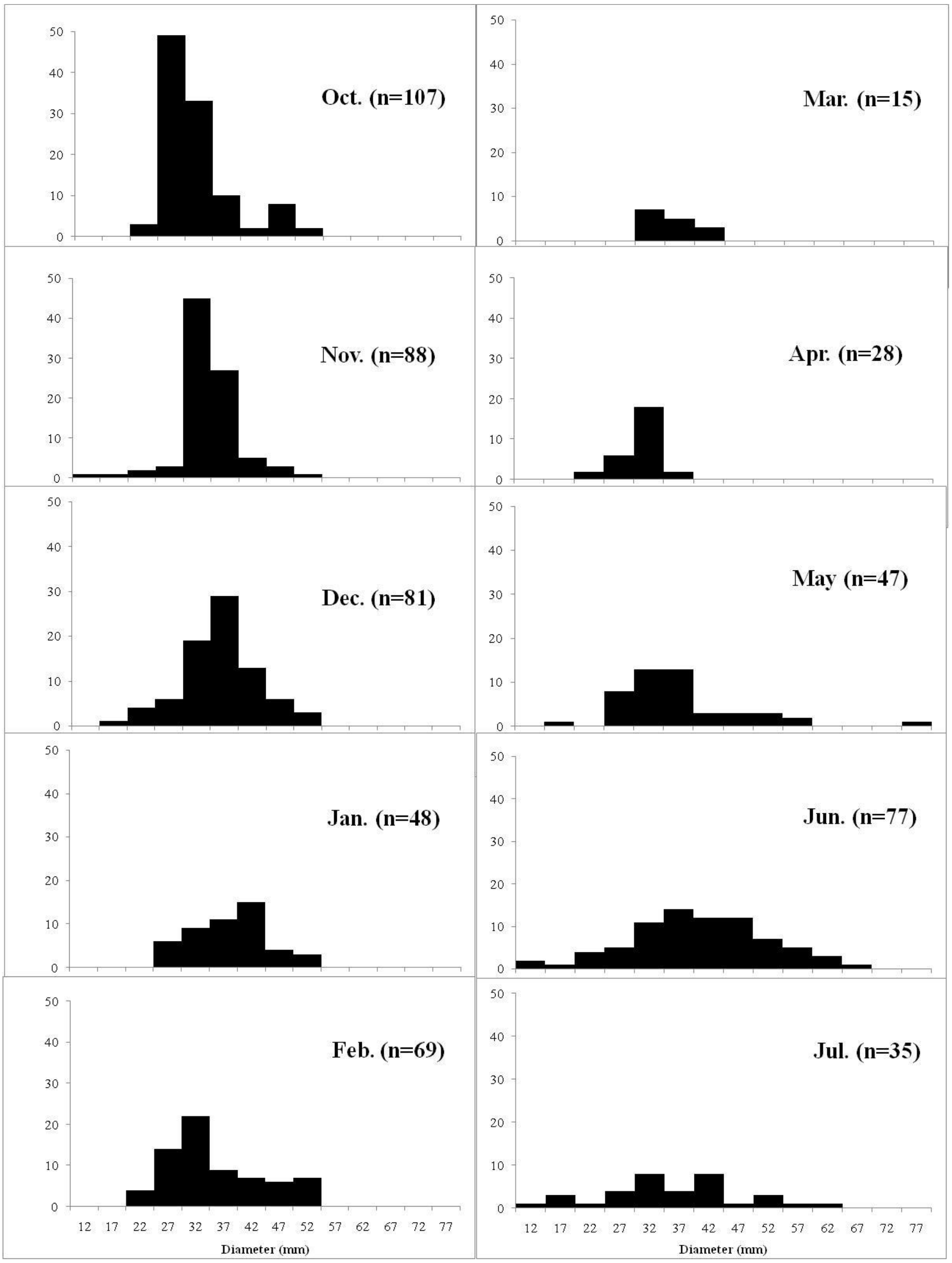

Figure 6.4. Monthly size structure of trochus collected at the intertidal area of Sabang Reef from October 2009 to July 2010.

Growth of translocated and non-translocated trochus

The general growth pattern of translocated and non-translocated trochus were declining and becoming less variable with increasing shell's diameter, but translocated trochus were either having a very fast or very slow growth rates than non-translocated individuals. The projected von Bertalanffy growth curves indicate that growth of trochus in Site 1 could be twice as fast as at Site 2 during the first year. Those in Site 3 had the slowest growth. The $L\infty$ of trochus in Site 1 and 2 were relatively close, but were lower than that in Site 3 (**Figure 6.5**).

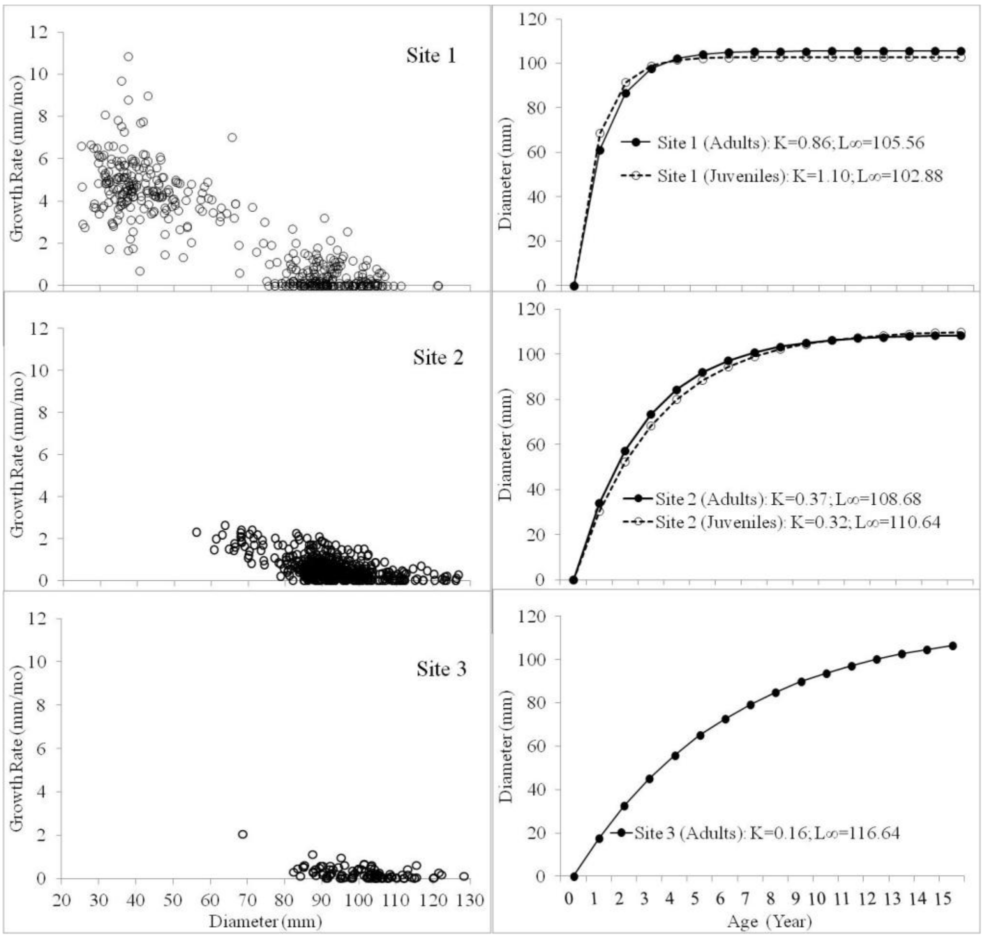

Figure 6.5. Variation in the monthly growth rates (mm mo^{-1}) of juveniles and adult trochus (left column) and projected diameters (mm) at age (year) estimated using the von Bertalanffy growth formula (right column) of translocated (Sites 1 and 3) and non-translocated trochus (Site 2) in Palawan, Philippines. The diameter (mm) at the x-axis of scatter plots (left column) is measured as the mid-point between the sizes at tagging and recapture or sizes at previous and succeeding recaptures.

Trochus in Site1 can reach a size of 60 mm during the first year while such size can be attained in two years at Site 2, and only after 4 years in Site 3. In three years time, 92 % of the shell's average maximum diameter is reached by trochus at Site 1 while only 67 % of the average maximum diameter is reached by trochus in Site 2. In Site 3, the predicted size at the age of 3 years comprised 39 % of the average maximum diameter.

Survival and recapture probabilities of translocated and non-translocated trochus

The survival and recapture probabilities of trochus varied according to size and site of translocation, with the survival rates of juveniles slightly lower than those of adults, and the recapture probabilities at Site 1 were much lower than in Sites 2 and 3 (**Figure 6.6**). The group 1 (juveniles and small adults) in Site 1 had time dependent survival and recapture probabilities (ϕtpt) (**Table 6.2**). The survival rates of juveniles increased with time which is normal as they grow larger, while their recapture probabilities were declining over time as they tended to move to shallow subtidal habitats as they grew bigger. For large trochus in Sites 2 and 3, survival and recapture probabilities were best explained by a combination of models. For the large trochus (group 2) in Site 1, survival probabilities were constant and with time dependent recapture probabilities (ϕ.pt). The recapture probabilities of adult trochus in Site 1 were also declining because they tended to move towards the deeper parts of the reef. Initially, the large trochus were released at the lowest water mark at low tide, but they gradually dispersed and most recaptured shells were spotted at 4 – 5 m deep. This downward movement was not noted for non-translocated trochus in Site 2 in spite of their gradually sloping habitat just like in Site 1. In Site 3, the translocated trochus did not move far from the point of release possibly because of reef and sand barriers surrounding the area of release.

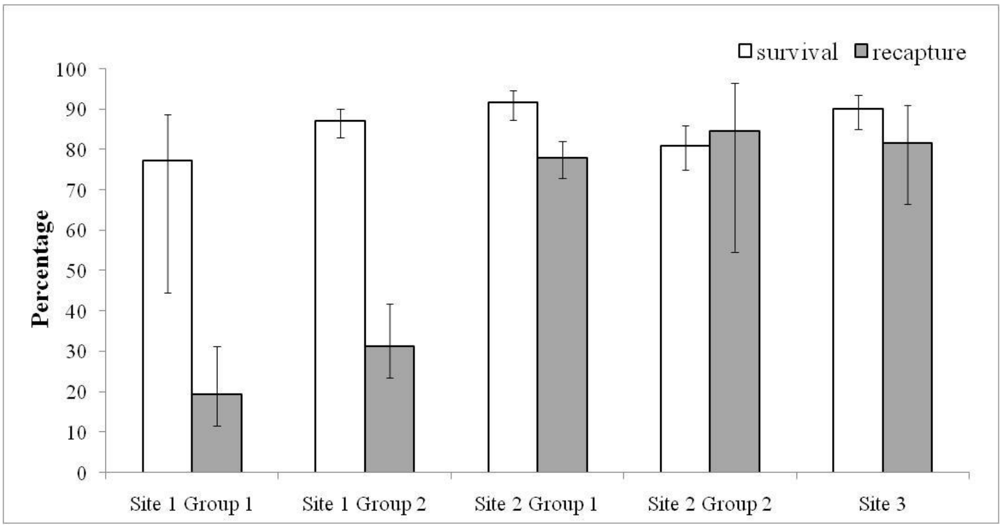

Figure 6.6. Averages of survival and recapture probabilities of translocated (Sites 1 and 3) and non-translocated trochus (Site 2) in Palawan, Philippines. The error bars are probabilities at 95 % confidence limits.

DISCUSSION

Growth of translocated and non-translocated trochus

The growth of translocated and non-translocated trochus varied between sites possibly due to abundance, type of habitat and availability of food. The growth rates of juveniles (about 40 mm diameter) at Site 1 as reflected in the scatter graph (**Figure 6.5, top left column)** ranged between 1 – 9 mm mo^{-1} or an average of about 5 mm mo^{-1}. The modal class frequency of the collected juveniles between October and January also suggested a 5 mm mo^{-1} growth rate. In other months, the modal trend was not visible possibly due to limited number of samples especially in summer (**Figure 6.4**). The result on growth of juveniles is suggesting that the place of translocation was a suitable habitat for trochus juveniles. The growth rates at this size were comparable to that of (Heslinga 1981; Hoang et al. 2007), and higher (0.8 – 4.2 mm mo^{-1}) compared to that in Guam (Smith 1987), and Vanuatu (> 2.3 mm mo^{-1}) (Purcell et al. 2004). As for the larger group, say for a 90 – 100 mm trochus had a growth rate ranging between 0 – 3.2 mm mo^{-1}, similar with those in Guam which ranged between > 0 – 3.0 mm (Smith 1987). Trochus of same size groups (30 mm and 90 – 100 mm) in Wallis Island had slower growth rates of 1.2 – 2.1 mm mo^{-1} and 0 – 1.4 mm mo^{-1} respectively (Lemouellic and Chauvet 2008).

The high variability in trochus growth (**Figure 6.7**) is not uncommon (Nash 1985; Smith 1987; Nash 1993; Purcell et al. 2004; Lemouellic and Chauvet 2008). Factors affecting variation in growth include salinity and temperature (Gimin and Lee 1997; Yi and Lee 1997), food and density (Nash 1985; Nash 1993; Lambrinidis et al. 1997; Hoang et al. 2007; Hoang et al. 2008). In this study, $L\infty$ was much smaller compared to other studies which could be the effect of the large number of small individuals with zero growth in the data.

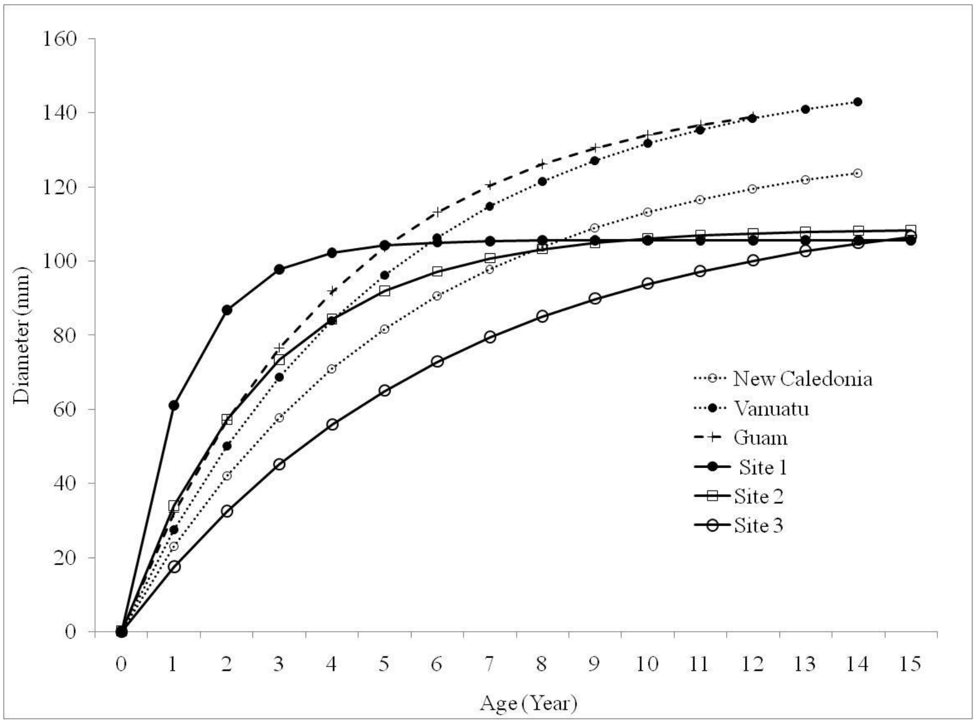

Figure 6.7. Estimated shell diameter (mm) at age (year) of translocated (Sites 1 and 3) and non-translocated trochus (Site 2) in Palawan, Philippines obtained using the von Bertalanffy growth formula compared with growth trends of wild trochus from Guam (Smith 1987), New Caledonia and Vanuatu (Nash 1993).

An abundance of food can make trochus grow to about 60 mm in one year even when under laboratory condition (Heslinga 1981). Other than food, saliniy and temperature can also affect the growth and survival of trochus. Gimin and Lee (1997) reported that the growth and survival of juveniles under laboratory condition was higher at a salinity of 30 ppt than at 35 and 40 ppt. Yi and Lee (1997) also suggested that in order to attain an optimum growth of captive juveniles, water temperature and salinity should be maintained at 30 – 31 ^{o}C and 31 – 37 ppt respectively. Low salinity was also reported to cause stunted growth among trochus near a river mouth, and where individuals grew larger when transfered to suitable areas (Nash 1993).

The faster growth of trochus at Site 1 compared to the two other sites could be attibuted to the favourable salinity levels (31 – 36 ppt) which was within the recommended range for juvenile trochus under laboratory conditions (Yi and Lee 1997) and low density which means more food would be available for the shells. During the rainy season, Site 1 was affected by river run off which drained nutrient rich water into the area, encouraged algal growth and helped maintain the optimum salinity for trochus growth. Water temperature fluctuated between 26 – 32 ^{o}C in all sites with a monthly average of about 29 – 30 ^{o}C which was also within the favourable growth requirement of trochus (Yi and Lee 1997).

Smith (1987) proposed that a narrow range of shell sizes in mark-recapture study can lead to some bias in estimates. In Sites 2 and 3, size ranges used were generally narrow, between > 50 to > 130 mm compared to that in Site 1. However, excluding individuals with size < 68 mm in growth analyses did not largely affect the model estimates at Sites 1 and 2 (**Figure 6.5**).

The reported sanility at TRNP, ranging between 34 – 35 ppt (Van Woesik 1996), was a little higher than in Site 1, and salinity could be a bit higher at shallow areas in summer time because of its isolation from the mainland Palawan. Such a high level of salinity could have slowed down the growth of the trochus at

TRNP as also noted by Gimin and Lee (1997) in their experiment. Another possible reason for slow growth could be limited food availability as a result of high density as in the case of Site 2, or limited food because of high sand deposition as we noted at Site 3. Although we did not measure the rate of sand deposits in the area, some giant clams at the shallow part of the reef were buried in sand, indicating heavy sand shifting on the reef slope inside the lagoon. Few trochus occur in calm areas with live corals and sandy substrate (Long et al. 1993), while suitable habitats adjacent to sandy areas do not support trochus populations (Colquhoun 2001). Schiel (1993) found that the major identifiable source of mortality of the seeded abalone was the partial or complete burial of the juvenile habitat associated with the shifting of sand.

Another evidence of extremely slow growths at Sites 2 and 3 was manifested by the presence of small juveniles (< 20 mm) in some areas at TRNP which do not have serrations on the base of the shell, while juveniles of the same size used in the mark-recapture study in Site 1 have serrated base (**Figure 6.8, top and bottom rows**). Among wild trochus, serrations disappear upon reaching the size of 25 – 30 mm which goes with a shift from cryptic to non cryptic behaviour, while hatchery-reared juveniles begin to produce a smooth shell whorls at 10 – 15 mm (Purcell 2002). Stunted trochus (< 20 mm diameter) reared in indoor tanks by WPU-SF for a year after settlement, still have some blunt serration (pers. obs.; **Figure 6.8, middle row**). This suggests that the < 20 mm trochus at TRNP with smooth edges and thicker shells had stunted growth and was therefore much older than trochus of the same size in Site 1. It may be even older than those one year old juveniles in the hatchery which still possess some blunt serration along edge of the shell's base. If the absence of serrations in small juveniles at TRNP indicates stunted growth, this supports the result of the von Bertalanffy growth model that trochus at TRNP were slow growing compared to those in Site 1. However, we do not disregard the possibility that fast growing juveniles may occur in other suitable areas of the reef within TRNP and also it could be possible that some stunted juveniles are occurring in some reef areas in mainland Palawan.

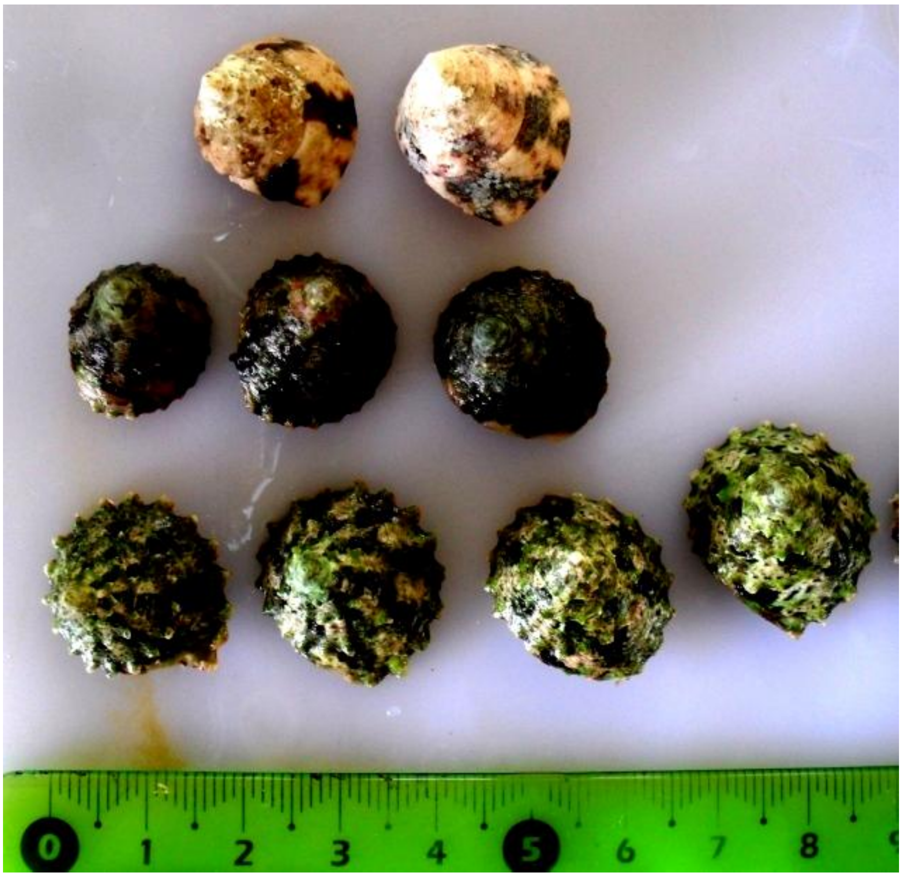

Figure 6.8. Stunted trochus at TRNP (top row) lacking the serrations that is normally found for smaller trochus in the hatchery (middle row) and in Site 1 (bottom row). Trochus in the middle row were one year old hatchery bred.

The growth rate of trochus in its habitat can influence its population growth. In Palau, trochus become sexually mature upon reaching a size of 50 mm in diameter in less than a year (Heslinga 1981) and the Japanese Hatchery (IMDC) had successfully induced to spawn 3-year old captive and stunted (~ 50 mm) trochus (**see Chapter 3**), although some authors have reported that sexual maturity can be attained upon reaching a size of 60 - 80 mm (see Nash 1993). If we assume that trochus in Palawan become sexually mature upon reaching a size of 50 mm irrespective of age, this means that in Site 1, it would take a year for a newly settled trochus to contribute in the natural recruitment, while it could be between 2 – 3 years in Sites 2 and 3. Given that growth is site specific and poor habitats can cause delay in sexual maturity, careful habitat selection is important to obtain fast population growth of translocated trochus. Well-protected habitats known to hold high populations of trochus could be a good choice for translocation or aggregation of breeding adults to hasten their population growth.

Survival and recapture probabilities of translocated and non-translocated trochus

The average survival probability of 0.77 (95 % CI: 0.44 – 0.89) of wild juveniles in this study was much higher than hatchery-produced juveniles released into the wild, while those of adults were comparable to other studies. For example Castell (1993) found that low surival rates of 5 – 14 mm juveniles due to predation occurred under laboratory trials, which she converted into a discouraging 25 (0.13 %) surviving individuals in 6 months out of 20 000 initial stock. A zero survival after 130 days from the date of release was reported by Isa et al. (1997) for juveniles measuring 8 – 16 mm. The tendency of the hatchery produced juveniles to wander compared to their cryptic wild counterparts make them vulnerable to predation (Purcell 2002). The mean survival probabilities of large trochus from Sites 1 – 3 ranging between 0.81 and 0.92 were comparable to 0.925 annual survival rates obtained from recovery of dead shells in a tagging experiment, but higher than 0.46 – 0.77 survival rates obtained using length-converted catch curve analyis (Nash 1993 and references therein).

The mean recapture probabilities (about 0.2) for juveniles in this study were lower compared to other studies. For example, Purcell et al. (2004) and Castell et al. (1996) reported higher probabilities of recapture ranging between 0.4 – 0.5 and 0.7 – 0.8 respectively. The variation in recapture rates could be related with the season and complexity of the habitat (Purcell and Cheng 2010), the interval between the release and recapture, and the sizes of the shells. Purcell et al. (2004) recaptured the shells (size range: 35 – 45 mm) after 6 months while Castell et al. (1996) did the recapture of 17 – 48 mm shells 2 – 3 days after the release. The intertidal area in Site 1 was characterised by large rocks which makes searching for shells difficult and could be the reason for low recaptures rates. Castell et al. (1996) suggested that a large proportion of shells are missed during the recapture and the proportion of recaptured individuals may vary with time and space.

The tendency of trochus to move considerable distances when introduced to a new habitat is important to consider in selecting a trochus conservation site. Smith (1987) explained that the low recovery for tagged trochus could be a result of migration, predation and harvesting. Rao (1936) lost his introduced trochus a month after the time of release which he attributed to the tendency of introduced trochus to wander in the new habitat. At Site 1, we ignored the effect of harvesting because of the prior arrangement with the community, so that those which were not recaptured were missed or may have moved to the deeper parts of the reef which were inaccessible through breath-hold dive, the method which we employed during the recapture. By contrast, recaptured trochus in Site 2 only moved few meters away from the area of release. Scuba diving at the deeper part of Site 2 did not reveal the presence of any tagged trochus although there were a few large untagged individuals. This suggests that those which we missed during the recapture in Site 2 were hiding in intricate coral crevices at the shallow subtidal area or had been consumed by predators. According to Nash (1985) trochus does not move great distances. His tagged individuals were found only about 30 m away from their point of release after two years. This may not be true for all trochus translocated to a new site as some large trochus at Site 1 were found about 100 m away from the point of release after 30 days. At Site 3, movement of translocated trochus was only within few meters from the point of release possibly because of coral walls and sand patches surrounding the release site. In this case, translocation of trochus in well-protected MPAs could be feasible only when suitable habitat is bordered with deep water or when there are reef structures that inhibit the translocated trochus from moving out of the MPAs.

Implications for conservation

The conservation of trochus in a country like the Philippines may be best achieved by increasing the use of translocation of wild-caught individuals, as this is likely to be more cost-effective and more successful than the release of mass-produced trochus juveniles. The absence of large trochus in a guarded reef slope near the hatchery which had been a release site of IMDC (**Chapter 3**) is a good illustration that the release of hatchery-produced juveniles is less successful than introducing wild-caught trochus. Most restocking projects involving the release of hatchery produced juveniles have failed to provide evidence of success (Isa et al. 1997; Crowe et al. 2002), in contrast to a number of successfully established populations obtained by translocation of wild adults to a new habitat (Sims 1984, Gillett 1993; Ponia et al. 1997; Purcell et al. 2004; Pakoa et al. 2010). Thus restocking of wild trochus is best accomplished by the translocation of wild individuals (Bell et al. (2005) if the habitats favour recruitment and growth and if the introduced trochus are not harvested over a longer period of time (Nash 1993). In trochus translocation, long-term monitoring of the sites of reintroduction and sites without intervention is suggested to determine whether the population have become established.

The high survival rates of wild juveniles in this study suggest the possible use of recruits for translocation, which are easier to transport in large numbers compared with large individuals. A small percentage (6 %) of young adults is collected along with juveniles in the wild. Given their high survivorship in the wild, the translocation wild juveniles would be more advantageous than sub-adult hatchery-produced juveniles which are still susceptible to high predation (Villanueva et al. 2010).

Trochus does not occur in suitable habitat near sandy areas (Colquhoun 2001) which means that careful habitat selection is important in trochus translocation sites, or in releasing confiscated trochus. The Rangers of TRNP noted high mortalities for confiscated trochus when released at intertidal sandy areas, although it was not certain whether mortalities were due to extreme temperature at low tide, predation, or stress of the collected animals. Trochus occur at rocky intertidal areas (Amos 1995; Colquhoun 2001) and can survive for long hours out of water (Crowe et al. 2002), so it may be more likely that the observed mortalities were associated with their vulnerability to predators when in sandy environments. Trochus need a hard surface to cling to, which sandy substrate does not provide, making them easily carried away by water current, buried by sand or turned up-side-down making them susceptible to predation.

The presence of recruits in heavily exploited Bitauran Reef, Sabang Reef and Rasa Island (**see also Chapter 2**) and in other parts of the country considering the inclusion of trochus in the catches of fishermen (Schoppe et al. 1998; Del Norte-Campos et al. 2003), suggests the possible presence of breeding population in deeper parts of the reef which the fishers cannot reach. However, a number of MPAs which have been protected for several decades seem not to have promoted the recovery of trochus. Most studies on MPAs focus only on fishes (Russ and Alcala 1996; Alcala 1998; Russ and Alcala 2003; Russ et al. 2004; Abesamis and Russ 2005; White et al. 2005). Very few have considered the effects of MPAs on molluscs such as giant clams *Tridacna gigas* (White 1986) and donkey’s ear abalone *Haliotis asinina* (Maliao et al. 2004). The breeding and restocking of invertebrates like giant clams in long-term managed MPAs (Cabaitan et al. 2008; Lebata-Ramos et al. 2010) further suggest that this species has not regained their former abundance. Raymundo (2003) assumed that the lack of giant clams, triton and pencil urchin in Apo Reef, Philippines, in spite of long-term protection, is due to recruitment limitation because adults are so rare on other reefs in the region, and that it is unlikely that these animals would regain their abundance. With the absence of empirical data on the state of trochus in many marine reserves in the country, stock assessment of this species and their potential habitats in other MPAs could be helpful in deciding the most appropriate and cost-effective means of conservation.

As the government is failing to effectively support the protection of large MPAs along with waning funding from external bodies (McClanahan 1999; Depondt and Green 2006; Thur 2010), the translocation and or aggregation of trochus within a network of village marine reserves could form part of a breeding population which could help augment recruitment in depleted areas. The tendency of recruitment within the parent population, and the limited movement of trochus after settlement (Nash 1993), makes them potentially suitable for small marine reserves. Empirical studies have shown that rates of recovery can be faster for relatively site-attached species (Russ and Alcala 1996), and the greatest rates of recovery are expected when species have relatively small home ranges and are non-migratory (Kramer and Chapman 1999). Also, the use of wild juveniles or adults could be an advantage as this would bypass the need for hatcheries and removes the risks of introducing genetically modified animals in the wild (Bell et al. 2006) and viral diseases (Kitada and Kishino 2006). At present, over 91 % of the > 600 MPAs in the Philippines are managed under the community-based coastal resource management (CBCRM) framework (Maliao et al. 2009), a management strategy which seems better at conserving trochus populations than previous approaches (ICECON 1997; Purcell 2004; Samoilys et al. 2007; Dumas et al. 2010; Hind et al. 2010; Pollnac et al. 2010). However, given the long wait between the time of reintroduction and the start of first commercial harvest (Sims 1984; Passfield 1997; Ponia et al. 1997), or period of recovery after fishing moratorium (Purcell 2004; Ramakrishna et al. 2010), good governance is important so that reintroduced trochus and their recruits are not harvested (Nash 1993; Purcell 2004). This requires the reintroduction of trochus to well-managed village MPAs or to privately-managed reserves (Svensson et al. 2008, Svensson et al. 2010). Studies involving transplantation of coral and restocking of giant clams in degraded reef areas have resulted to a significant increase in fish abundance (Cabaitan et al. 2008). The restoration of trochus population along with other macro-algal grazing species in a well-managed coral reef can help promote the health of the reef, enhance its capacity to support a diverse fauna, increase its resilience against climate change, and sustain the goods and services that people need from the reefs (see Moberg and Folke 1999).

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Chapter 7

THE USE OF ELASTICITY ANALYSIS ON AGE-BASED MATRIX MODEL IN PROPOSING CONSERVATION MEASURES FOR *Trochus niloticus*

Roger G. Dolorosa, Alastair Grant and Jennifer A. Gill

ABSTRACT

Elasticity analysis is an important tool for determining which life stages of commercially important and threatened species should be the focus of conservation. Elasticity analyses were conducted for a deterministic age-based matrix model of unexploited and heavily fished slow and fast growing populations of the reef gastropod *Trochus niloticus* to determine the age classes making the highest contribution to intrinsic population growth (λ). Adult survival rates were determined by fitting mixture models to population size structures. The juvenile survivals from egg up to one year old (P_o) required to have a stable (λ = 1) and growing (λ = > 1) population sizes were estimated by iteration. In the matrix model, different levels of adult survival rates (P_i) were set constant for all age groups in the unexploited and heavily exploited populations. Adult survival rate ranged between 59 – 85 % in unexploited population. Juvenile survival rates required to promote a stable (λ = 1) and growing population (λ = 2.4) were at least 15 ind and 280 ind for every 100 million eggs. For heavily exploited population, at least 212 ind per 100 million eggs are required to reach one year of age to have a stable population. Perturbation analyses showed that survival elasticities were much larger than fecundity elasticities for unexploited trochus populations, with both survival and fecundity elasticities declining with increasing age. Under heavy fishing and when λ = 1, survival elasticities were lower than fecundity elasticities for both slow and fast growing trochus. For both slow and fast growing trochus under heavy fishing condition, only the first 3 age classes have elasticity values greater than zero, which suggests that increasing the survival of wild sub- and young adults would have a greater effect on population growth, while conservation focusing only either on the protection of juveniles or very large adults is of little benefit as a management strategy and will not ensure the recovery of the trochus population.

Keywords: elasticity, matrix model, population growth, *Trochus niloticus*

INTRODUCTION

In population management, it is important to identify the most effective focus for management effort, whether protecting the life history stage that will contribute most to the population growth of a threatened species or targeting vulnerable life stages of pest species (Benton and Grant 1999; Govindarajulu et al. 2005). Perturbation analyses such as elasticity and sensitivity analyses provide information on which life stages of the animal have the greatest effects on a population level (Benton and Grant 1999; Caswell 2000; Heppell et al. 2000). They are widely used in predicting the changes in fished populations and in managing populations of endangered species (Crouse et al. 1987; Donovan and Welden 2002; Rogers-Bennett and Leaf 2006) or in controlling unwanted species (Benton and Grant 1999; Caswell 2001; Govindarajulu et al. 2005; Hansen 2007). These analyses have become an important tool in the management of marine turtles (Crouse et al. 1987), otters (Gerber et al. 2004a; Gerber et al. 2004b), fishes (Pfister 1996; Armsworth 2002), some coral species (Hughes and Jason 2000), some molluscs (Barbeau and Caswell 1999; Rogers-Bennett and Rogers 2006; Rogers-Bennett and Leaf 2006), and a variety of other marine species (Benton and Grant 1999; Gerber and Heppell 2004).

In conserving endangered species, many programmes have focused on increasing the survival of the early life stages such as eggs and juveniles, so called "headstarting", but were later shifted to the protection of adults following the conduct of perturbation analyses on the target populations. For example, in the case of Loggerhead sea turtles *Carreta carreta*, the production of juveniles was the focus of conservation efforts for 15 years, but was subsequently found to be ineffective in halting population decline. Compiling and analysing life tables of this marine turtle species indicated that protection of adults would have a greater probability of reversing the population decline. The result was a shift of management efforts to the protection of adult turtles through the installation of turtle exclusion devices on fishing nets (Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1996; Donovan and Welden 2002). Life history analysis on black bear populations also suggests that adult survival should be the primary target of conservation and management strategies (Freedman et al. 2003). For white abalone *Haliotis sorenseni*, the largest adults were found to have the most influence on population growth (Rogers-Bennett and Leaf 2006). For sooty shearwater birds *Puffinus griseus*, the harvesting of adults had negative effects on population growth 10-fold greater than the effect of chick harvesting (Hunter and Caswell 2005). For the southern sea otters (*Enhydra lutris*), elasticity analysis indicated that populations were highly sensitive to changes in adult survival rates (Gerber et al. 2004a; Gerber et al. 2004b).

It is however difficult to generalise that only the protection of adults would promote a higher possibility in conservation success. Heppell et al. (2000) found contrasting patterns of elasticities among slow and fast growing mammals. The former have the tendency to breed at later years and with few offspring were found to respond better to improved adult or juvenile survival rates, while the latter responded to improved survival rates of offspring. For the red (*Haliotis rufescens*) and white (*H. sorenseni*) abalone, Rogers-Bennett and Leaf (2006) suggested a different focus on conservation between the two species after conducting an elasticity analyses on both populations. Red abalone population would respond well to conservation when sublegal sizes (150 – 178 mm) are protected from harvesting, while for white abalone, the largest size-class (140 – 175 mm) have the highest elasticity and that conservation effort should be directed towards reducing mortality of these size group.

A conservation method involving the release of mass produced juveniles in an effort to rebuild depleted stocks and increase capture fisheries catches has been successfully carried out for several decades in a number of commercially important species (Burton et al. 2001; Mustafa et al. 2003; Hamasaki and Kitada 2006; Kitada and Kishino 2006; Wang et al. 2006; Hamasaki et al. 2011), but in general, stock enhancement has failed to significantly increase global fisheries production (Lorenzen 2008). Stock enhancement strategy has been successful for Japanese scallop *Patinopecten yessoensis* in Hokkaido, Japan but were not successful in other places, except in the South Island, New Zealand (Bell et al. 2006; Uki 2006). In a stock enhancement site in Japan, catches of the released hatchery-produced red sea bream *Pagrus major* and Japanese flounder *Paralichthys olivaceus* have contributed about 36 % and 23 % of the total catches for each species. However releasing millions of hatchery-produced juveniles could be insignificant compared to natural recruits produced by the large numbers of natural stocks in Japan (Kitada and Kishino 2006). Stock enhancement of kuruma prawn, *Penaeus japonicas,* in Japan for more than 30 years since 1964 involved the release of ~ 140 – 300 million juveniles per year. Its success was dependent on availability of tidal nursery habitats but in general it has failed to augment total production (Hamasaki and Kitada 2006). Mass releases of around 600 million juveniles per year (and more than 5 billion in 1991) of the prawn *Penaeus chinensis* for more than 20 years in China have augmented the production of commercial prawn fisheries where catches of released prawn contributed > 90% of the total landings in one region but overfishing has been presumed to have prevented the increase in size of wild population (Wang et al. 2006). So in spite of some success stories, stock enhancement in general has failed to significantly increase global fisheries production.

Trochus is a reef gastropod which has been heavily exploited mainly for the production of mother-of-pearl buttons. One of the many conservation efforts to rebuild their populations is the release of hatchery produced juveniles (Nash 1993; Purcell 2004) but this has met with limited success (Isa et al. 1997a; Purcell 2001; Crowe et al. 2002; Purcell et al. 2004; Purcell and Cheng 2010). Nash (1988; 1993) and Bell et al. (2005) pointed out that the feasibility of juvenile re-seeding as a management tool depends on high survival between seeding and harvesting, and economical viability or production costs before releasing. To evaluate whether “headstarting” is an appropriate means of trochus conservation, we conducted a prospective elasticity analyses to determine which age classes of trochus have the greatest contribution to intrinsic population growth in a deterministic age-structured matrix model. Specifically, we determined the following: a) survival probabilities of adults; b) juvenile survival required for exploited and unexploited populations to have a stable and increasing population growth, and c) fecundity and survival elasticities of slow and fast growing groups of trochus when the population is either protected or heavily exploited in the deterministic age-structured matrix model.

METHODS

This study which mainly aimed to determine the age classes of trochus with the highest contribution to population growth, made use of the following four models: Mixture, Cormack-Jolly-Seber (CJS), regression and deterministic age-structured matrix models. The Mixture model was used to determine the survivorship of trochus using size frequency distribution data. The CJS model was also used to determine the survival probabilities of trochus but data from mark-recapture were used. Regression analysis was used to predict the fecundity at age at particular size of trochus used in the age-structured model. And the deterministic age-structured matrix model was used to determine the survival and fecundity elasticities of different age classes of trochus to changes in the vital rates such as growth and survival rates.

Annual survival probabilities of *Trochus niloticus*

The annual survival probabilities of trochus were determined from mark recapture data using the Cormack-Jolly-Seber (CJS) model (**see Chapter 6**) and by fitting mixture models to size structure data. Both models were used to estimate the annual survival rates of trochus in Tubbataha Reefs Natural Park. Only the CJS model was used for trochus from the mainland Palawan because too few individuals were sampled to obtain data on population size structure.

In the mark-recapture method (**see Chapter 6**), survival probabilities of five groups of trochus were monitored for 160 – 280 days. Sampling was conducted monthly in Binduyan and every 2 – 4 months in TRNP. In total 1 251 trochus were marked with 1 126 individuals recaptured in 26 samplings or recapture events (**Table 7.1**). Recapture of trochus for each group per site was restricted to 1 – 2 hours during each sampling event. The survivorship probabilities for slow growing trochus in TRNP and for fast growing trochus in Binduyan was determined using the CJS Model (**see Chapter 6**) within the Program Mark (White and Burnham 1999; Cooch and White 2010). The average survival rates obtained for juveniles plus a few small adults were 77 % (95 % CI: 44 – 79 %), while the mean survival probabilities of large trochus ranged between 81 % and 92 % (**Table 7.1).**

Table 7.1. Mean survival rate of trochus in TRNP (slow growing) and Binduyan (fast growing) obtained through mark-recapture method (**see Chapter 6**). TR – total number of recaptured individuals; RE – number of recapture events. *Juveniles with about 6 % small adults.

Site	Group	Duration (days)	Mean Size (mm) ±sd	TR	RE	Survival Rate (%)	
						Mean (mm)	Confidence Limit (95 %)
TRNP (slow growing)	1 (n = 403)	242	93.5 ± 11.0	473	3	92	87-95
	2 (n = 83)	164	82.0 ± 18.7	72	2	81	75-86
	3 (n = 108)	184	99.9 ± 10.9	108	2	90	85-94
Binduyan (fast growing)	1 (n = 532)*	278	34.5 ± 8.2	237	10	77	44-89
	2 (n = 125)	270	90.6 ± 10.9	236	9	87	83-90
Total	1 251			1 126			

In the mixture model, the Mixdist package in R was used to find a set of overlapping component distributions that gives the best fit to grouped size frequency data using a combination of Newton-type iterations and the Expectation Maximization (EM) algorithm (Macdonald 2003). In doing this, we used the raw data on sizes of trochus (n = 1 219, mean = 66.9 mm, sd = 17.4) obtained from TRNP in 2006 prior to the commencement of heavy poaching (see Dolorosa et al. 2010; Jontilla et al. 2011; Jontilla et al. in press) and the predicted mean size at age from mark-recapture experiments which were obtained using the von Bertalanffy growth formula (Ogle 2010; **see Chapter 6**). We then fix the starting values for parameters of a mixture model using the predicted mean size at age (obtained using mark-recapture study) and an assumed standard deviations of 20 mm at each age class except in age 1 in which we used 15 mm (assuming that sizes of juveniles are less variable than adults), before coding the R program (R 2011) to fit a mixture model for both normal and lognormal distribution. The predicted proportion at each age class in the mixture model was used to estimate the survival at certain age group by dividing the proportion of the next higher age group with the proportion of the lower age group. Estimating the demographic parameters from size-frequency data by constraining some parameters to particular values can provide important information even when the modes at the size frequency are not well separated (Grant et al. 1987).

The estimates obtained from the mark-recapture and mixture models were used as bases for deciding the survival rates of adult trochus to be used in the model under no fishing condition.

Fecundity of *Trochus niloticus*

The fecundities or number of eggs that a mature female trochus of particular size or age from TRNP or Binduyan can spawn was based on the fecundity data of Bour (1989). The fecundity data of Bour (1989) were plotted as a scatter graph against size and an exponential regression fitted. Trochus can reproduce 3 times in two years (Hahn 1993), so predicted annual fecundity for the mean size at each age was multiplied by 1.5 and the products rounded to the nearest 100 000.

The age-structured matrix model

We used a deterministic age-structured matrix model (Jensen 1974; Groenendael et al. 1988; Caswell 2001; Donovan and Welden 2002) with 13 age classes for both slow and fast growing trochus**.** The use of age-structured models is common for studies on birds and mammals where development is more deterministic (Groenendael et al. 1988; Jensen 2000; Jensen and Miller 2001). For organisms with markedly plastic development, the use of stage-structured model instead of age categories are superior because demographic fate is more closely predictable from stage classes than from age categories, however, age may be important in addition to size even in organisms with plastic development (Groenendael et al. 1988, and references therein). Trochus exhibit plastic growth which varies between regions (Nash 1993), and for the case of the studied trochus, growth at TRNP was slower than at Binduyan (**Chapter 6**), so these two groups of trochus were modelled separately. Juvenile survival, and adult survival and fecundity elasticities were estimated in a projection matrix with age specific fertilities (F_{1-13}) arranged on the first row, age-specific survival probabilities (P_{1-12}) on the sub-diagonal row, and zeros in other parts of the matrix:

$$\mathbf{A} \quad = \quad \begin{bmatrix} \boldsymbol{F_1} & \boldsymbol{F_2} & \boldsymbol{F_3}\ldots & \boldsymbol{F_{11}} & \boldsymbol{F_{12}} & \boldsymbol{F_{13}} \\ \boldsymbol{P_1} & 0 & 0 & 0 & 0 & 0 \\ 0 & \boldsymbol{P_2} & 0 & 0 & 0 & 0 \\ 0 & 0 & \boldsymbol{P_3}\ldots & 0 & 0 & 0 \\ 0 & 0 & 0 & \boldsymbol{P_{10}} & \mathbf{0} & 0 \\ 0 & 0 & 0 & 0 & P_{11} & \mathbf{0} \\ 0 & 0 & 0 & 0 & 0 & \boldsymbol{P_{12}} \end{bmatrix}$$

Estimating juvenile survival (P_o)

For most invertebrates, the survival from eggs to one year old (P_o) in the wild is not known (Rogers-Bennett and Leaf 2006). This is also the case for trochus so we estimated the value of P_o that would result in a stable population ($\lambda = 1$) by iteration of the deterministic age-structured Leslie matrix model

(Groenendael et al. 1988; Caswell 2001) composed of 13 age classes. In the matrix, the Fertilities (F_i) were obtained by dividing the annual fecundities by two on the assumption that only 50 % of the eggs will hatch as females, because studies have shown an equal ratio between sexually matured male and females (Gimin and Lee 1997; Pradina et al. 1997). The juvenile survival (P_o) was determined at each of the following conditions: a fast or slow growing population where adult survival (P_{1-12}) was held constant at different levels such as 75, 82 and 90 % for unexploited and 10 % for heavily exploited conditions in the model. A juvenile survival probability of 0.00000000005 or 5 individuals will reach one year old for every 100 000 000 000 eggs was used as starting number, then multiplied this repeatedly by 1.5 up to 0.000009 or 9 individuals will reach 1-year old for every 1 000 000 eggs, and calculated population growth rate (λ) for each. The relationship between juvenile survival and λ were plotted and these were used to determine the juvenile survival probability corresponding to a stable population. The possible maximum juvenile survival from a newly established population was estimated based on the intrinsic population growth (λ) of the translocated trochus. In this case, λ was obtained by dividing the recent population estimate with the initial number of stock raise to the power of 1/t, where t = number of years. The estimated intrinsic population growth for the 439 individuals firstly introduced in Tongavera that reached about 27 300 individuals after 10 years (Ponia et al. 1997) is 1.5. In Tahiti, trochus population was estimated to have reached 10 million 14 years after the introduction of 40 individuals (Cheneson 1997) thus λ = 2.4. The software R version 2.12.0 was used to estimate the juvenile survival (R 2011).

Elasticity analyses for slow and fast growing *Trochus niloticus*

Elasticity analysis is used to "estimate the effect of a proportional change in the vital rates on population growth" (Donovan and Welden 2002). In this study, we performed the elasticity analyses by fixing the survival probabilities from age 1 to 12 years (P_{1-12}) constant at 75, 82 or 90 % for both slow and fast growing population of unexploited trochus in the wild. For heavily exploited condition, we assumed a constant survival probability of 10 % for all age classes (P_{1-12}) because of indiscriminate harvesting of all size classes of trochus. Fertility values (F_i) at each age class was obtained by taking only half of the annual fecundity values per age class on the basis that 50 % of the eggs will hatch as females, multiplied by the corresponding juvenile survival (P_o). The elasticity analyses was performed using PopTools (Hood 2010). In our mark-recapture study (**Chapter 6**) we found that survival of trochus can be time independent so in the age structure model we maintained a constant survival rate for all adults. Similar patterns of annual survival have been reported for different stages of long-lived breeding loggerhead sea turtles *Carretta carretta* (see Crouse et al. 1987).

RESULTS

Survival probability of *Trochus niloticus* obtained by mixture model

Using size at age data from the von Bertalanffy growth model, the classification of individuals into age classes based on fitting normal and lognormal components in the mixture model showed little difference between models (**Figure 7.1).** The mean survival probabilities obtained using both fits were > 0.70. The underrepresented number of trochus in age class 1 precluded the estimation of survival for age class 2 (**Table 7.2**).

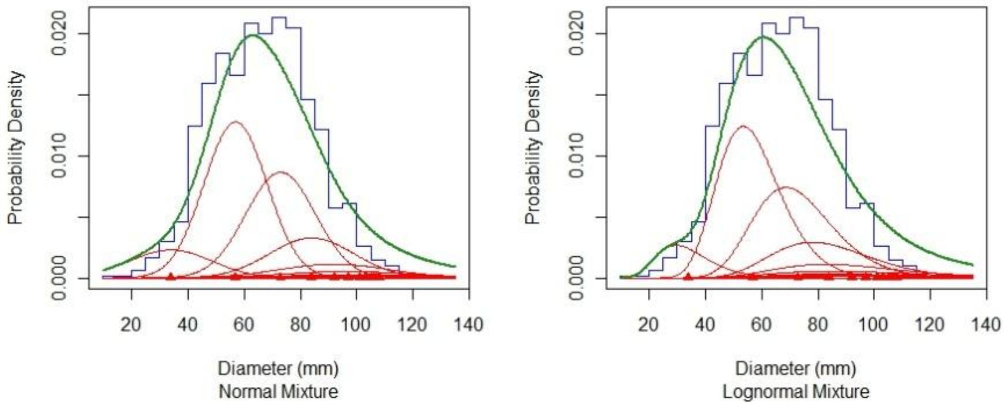

Figure 7.1. Normal and lognormal size at age segregation of slow growing trochus obtained by mixture model.

Table 7.2. Survival probabilities of trochus in TRNP at each age class obtained by using normal and lognormal fits in the mixture model. The mean sizes at age estimates were obtained from mark-recapture study using the von Bertalanffy Growth model (**Chapter 6**). The proportion of the sample belonging to 1-year old is under represented precluding the estimation of survival for 2-year old trochus. No survival is assumed at age 13.

		Normal fit			Lognormal fit		
Age (year)	**Mean size (mm)**	**Proportion per age class**	**sd (mm)**	**Survival probability**	**Proportion per age class**	**sd (mm)**	**Survival probability**
1	34	0.083	14.56		0.063	10.53	
2	57	0.358	11.17		0.350	11.84	
3	73	0.268	12.34	0.75	0.263	14.90	0.75
4	84	0.124	15.04	0.46	0.130	18.87	0.49
5	92	0.056	19.70	0.45	0.063	24.62	0.48
6	97	0.032	23.91	0.57	0.037	29.61	0.59
7	101	0.02	27.82	0.63	0.024	34.30	0.65
8	103	0.016	29.90	0.80	0.019	36.86	0.79
9	105	0.012	32.05	0.75	0.015	39.53	0.79
10	106.1	0.011	33.25	0.92	0.013	41.06	0.87
11	106.9	0.00994	34.13	0.90	0.012	42.19	0.92
12	107.8	0.00924	34.80	0.93	0.011	43.04	0.92
Mean				0.72			0.73
Lower limit confidence interval (95 %)				0.59			0.61
Upper limit confidence interval (95 %)				0.85			0.84

Fecundity of *Trochus niloticus*

The regression equation obtained from the data of Bour (1989) is $y = 138.48e^{0.0218x}$, ($R^2 = 0.9597$), where, y = number of eggs; x = size of shells in mm. The fecundity data of Bour (1989) and the fecundity estimates for slow and fast growing trochus obtained using the derived equation is reflected in **Figure 7.2.**

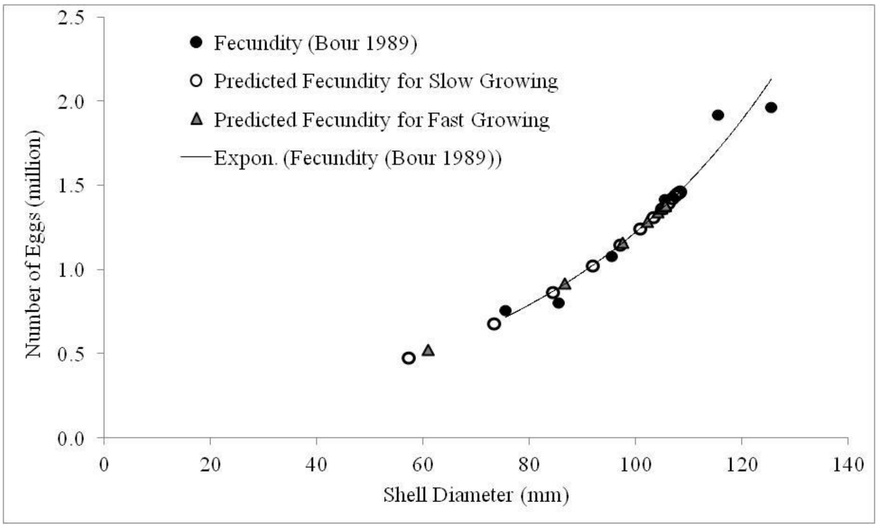

Figure 7.2. Predicted fecundity of slow and fast growing trochus in Palawan, Philippines obtained using the derived equation: $y = 138.48e^{0.0218x}$, ($R^2 = 0.9597$) from the fecundity data of Bour (1989).

The fecundities for the two model populations are on **Table 7.3.** Slow growing trochus attained sexual maturity size at the age of 2 years, while the fast growing trochus were assumed to have started to reproduce upon reaching 1-year old. The estimated number of eggs per year that trochus can release ranged between 0.72 – 2.18 million. However, slow growing groups can only produce a large number of eggs at later age (a 6-year old trochus can produce 1.7 million eggs per year) whereas in the fast growing group, a 3-year old trochus can produce about 1.7 million eggs per year.

Table 7.3. Predicted size at age of slow and fast growing trochus obtained using the von Bertalanffy Growth formula (**Chapter 6**) and the estimated number of eggs each trochus at a particular age or size group can release per year.

Age (y)	Slow growing		Fast growing	
	Size (mm)	Fecundity y^{-1} ('000)	Size (mm)	Fecundity y^{-1} ('000)
0	0	0	0	0
1	33.90	0	61.00	785
2	57.22	723	86.75	1 377
3	73.27	1 026	97.62	1 745
4	84.31	1 305	102.21	1 928
5	91.91	1 541	104.15	2 011
6	97.14	1 727	104.96	2 048
7	100.74	1 867	105.31	2 063
8	103.22	1 971	105.45	2 070
9	104.92	2 046	105.52	2 072
10	106.09	2 099	105.54	2 073
11	106.90	2 136	105.55	2 074
12	107.46	2 162	105.56	2 074
13	107.84	2 180	105.56	2 074

Juvenile survival and population growth

For the unexploited population where adult trochus are assumed to have a constant survival of 75, 82 and 90 %, the λ for both slow and fast growing population are of the same value and were increasing slowly as juvenile survival probability reaches $1 \times 10^{-6.}$ However, when juvenile survival probabilities were $> 1 \times 10^{-6}$ and when λ were either 1 or > 1, the values of λ for fast growing trochus increased rapidly, and when the juvenile survival was 9.59×10^{-6} , λ was more than twice as large as in slow growing trochus. Under heavy exploitation, the λ of both fast and slow growing population began to differ substantially when juvenile survival probabilities reached $> 1 \times 10^{-6}$. At a juvenile survival probability of 9.59×10^{-6}, the population growth rate of heavily exploited fast growing trochus was almost six times greater than that of slow growing trochus (**Figure 7.3**).

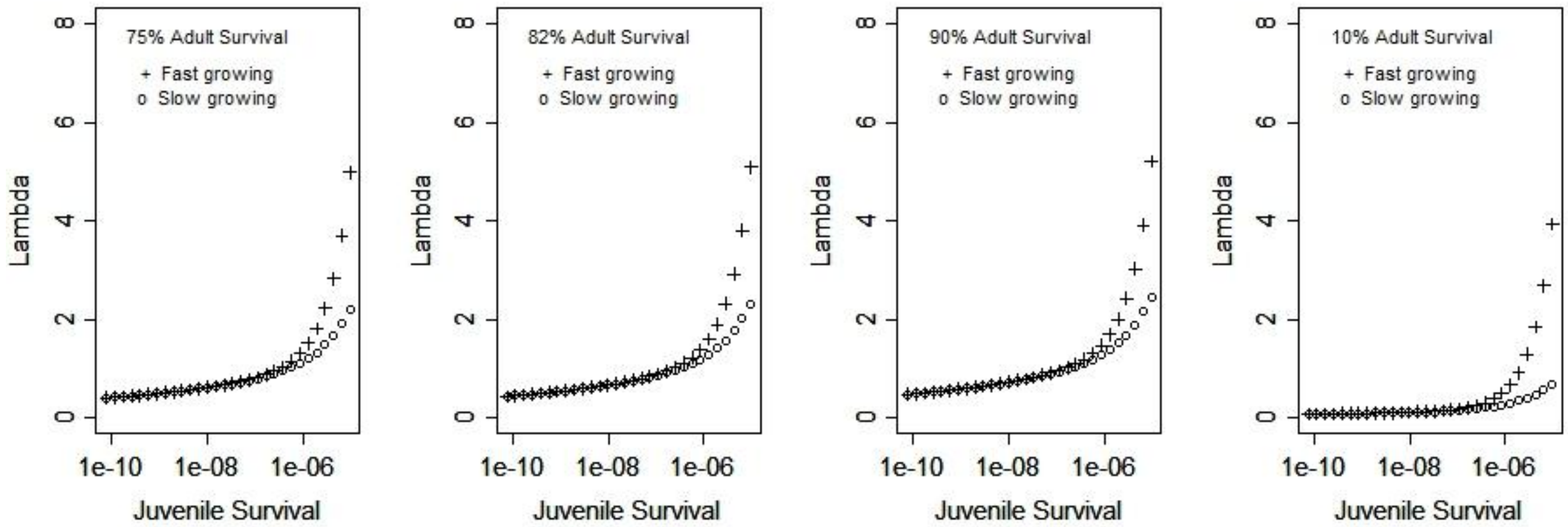

Figure 7.3. Predicted juvenile survival (P_o) in relation to population growth rate (λ) for slow and fast growing population at different levels (75, 82, 90 % for unexploited and 10 % for overly exploited) of adult survival (P_{1-12}) at every 1.5 step increase in juvenile survival.

The specific juvenile survival (P_o) that resulted to a stable population growth (**Figure 7.4),** indicated that when P_{1-12} was set at 75, 82 and 90 %, at least 53, 34 and 20 individuals for every 100 million eggs for slow growing trochus were required to reach the age of one year old so that the population was in a stable condition ($\lambda = 1$). For fast growing trochus, the required numbers of juveniles to reach one year old (P_o) so that $\lambda = 1$, were 33, 24 and 15 individuals for every 100 million eggs. These numbers are about 25 – 40 % lower than what is required for slow growing group.

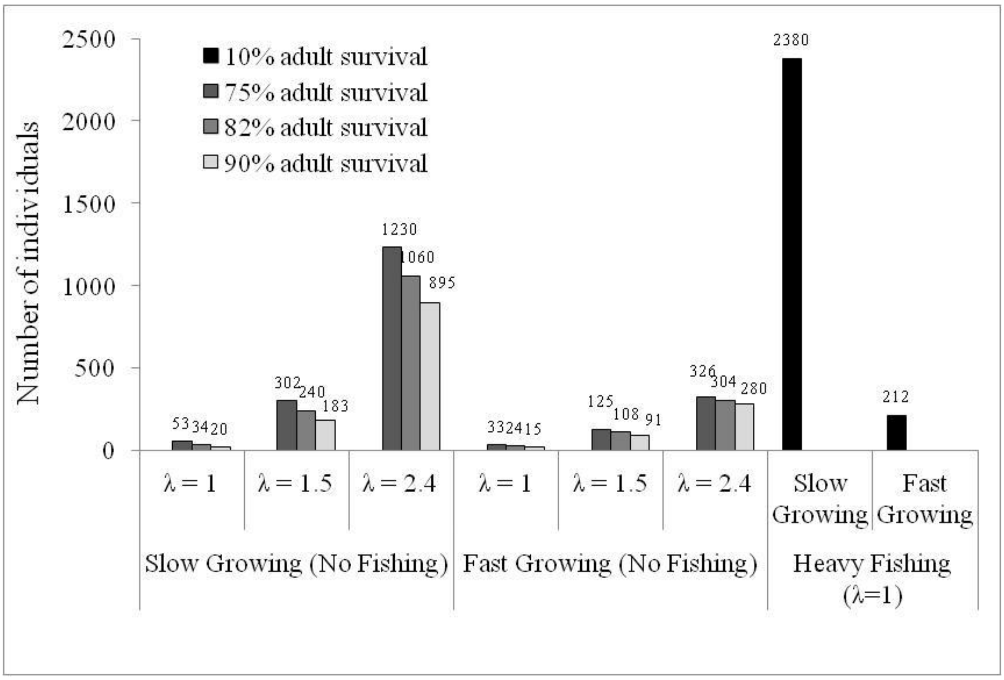

Figure 7.4. Estimated number of juveniles that reached 1 year old for every 100 million eggs at different levels of adult survival and population growth rates (λ) for slow and fast growing trochus either at no fishing or with high fishing conditions in the model.

For a newly established population having $\lambda = 2.4$ as for the case of the introduced trochus in Tahiti, the estimated number of juveniles that can reach the age of one year old for every 100 million eggs (when adult survival was fixed constant at either 75, 82 and 90 % in the model) could range between 895 – 1 230 individuals for slow growing trochus. These estimates are about 3.2 – 3.8 times higher than the estimated juvenile survival (280 – 326 ind of every

100 million eggs) of fast growing group having $\lambda = 2.4$. The estimated juvenile survival of a population with $\lambda = 2.4$ was about 3 – 5 times higher than the estimated juvenile survival for introduced trochus with a population growth rate of 1.5. Under extreme fishing condition (P_{1-12} = 10 %), the required number of juvenile survival for slow growing group was 2 380 individuals per 100 million eggs, while a much lower 212 individuals per 100 million eggs was required for fast growing trochus to have a stable population growth.

Elasticities of unexploited slow and fast growing *Trochus niloticus*

The survival elasticities for slow and fast growing trochus were much higher than the fecundity elasticities, but both declined with age (**Figure 7.5**). The declining trend is quite clear for survival elasticities because of large difference between the values at the smallest and largest age classes. About 52 and 56 % of the survival elasticities occurred within the first 3 age classes for slow and fast growing populations respectively. Survival elasticities on smaller age classes increased when adult survival probabilities (P_i) were reduced from 90 to 75 %.

The fecundity elasticities for both slow and fast growing trochus appeared relatively flat compared to the gradually sloping survival elasticities. The highest fecundity elasticities did not occur at the smallest age class, but was found at the next higher age classes, either at age classes 3 – 4 for slow growing or age classes 2 – 3 years for fast growing group. A gradual decline in P_i (from 90 to 75 %) resulted to a gradual increase in fecundity elasticities in favour of the smaller age classes and a gradual decline at the larger age classes. For example, the fecundity elasticities when P_i = 75 % were higher at the first 6 – 7 ages classes, but not at age classes 8 – 12 (**Figure 7.5**).

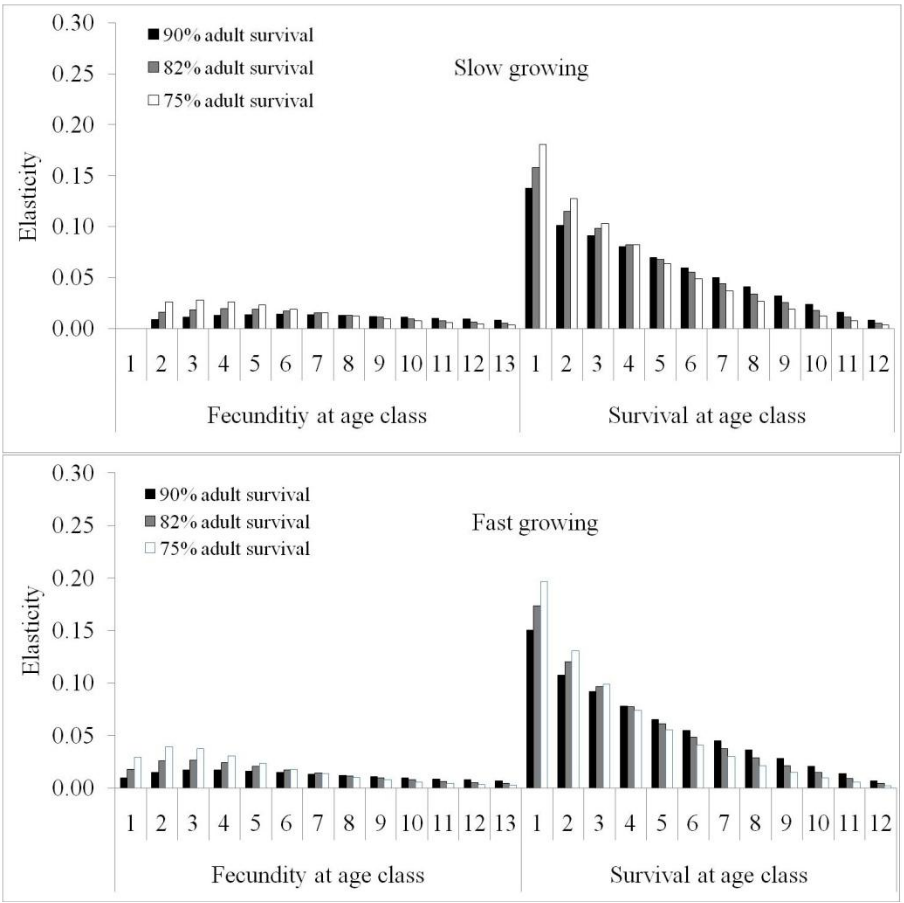

Figure 7.5. Fecundity and survival elasticities at different age classes of slow (top) and fast growing (bottom) trochus at different survival probabilities (P_i) and when $\lambda = 1$.

Elasticities of fast and slow growing *Trochus niloticus* under extreme fishing pressure

Under extreme fishing condition (P_i = 10 %), both fecundity and survival elasticities showed a rapidly declining pattern and were negligible for age classes 4 and above. The fecundity elasticities for both slow and fast growing group were much higher than survival elasticities. For 1-year old slow growing trochus, there was no fecundity elasticity value because the shells in the model measures < 50 mm and were considered immature. Among 2-year old slow growing trochus fecundity elasticity was about 3 times higher than survival elasticity. For 1-year old fast growing trochus, fecundity elasticity was about 5 times higher than survival elasticity (**Figure 7.6**).

Figure 7.6. Fecundity and survival elasticities of fast and slow growing trochus at 10 % adult survival as an effect of heavy fishing and when $\lambda = 1$. Note the absence of fecundity elasticity for immature 1-year old slow growing trochus.

DISCUSSION

The mean adult survival probability of trochus obtained with the mixture model (**Table 7.2**) is comparable to the mean values obtained from mark-recapture studies that ranged between 81 – 92 % (CL at 95 %: 75 – 95 %; **Chapter 6)**. In other studies, the annual survival rate of trochus can range between 44 to 77 % (Nash 1985) or even as high as 92.5 % (see Nash 1993). These findings indicate that trochus are long-lived repeat breeding animals and support the results of growth and age estimates using mark-recapture method that trochus can live for more than 10 years (Smith 1987; Nash 1993; **Chapter 6**). This further implies that large breeding adults have a vital role in maintaining a stable population growth.

For unexploited population to have a stable growth ($\lambda = 1$), juvenile survival of about 33 or 53 individuals (for fast and slow growing trochus) per 100 million eggs (at 75 % adult survival) is required (**Figure 7.4**). As the fecundity of a large female is about 2 million, (**Table 7.3**) a stable population requires only one of these eggs to develop and survive through to being a one year old juvenile for a stable population. The introduction of 40 mixed sex individuals in Tahiti which have reached an estimated population of 10 million individuals after 14 years (Cheneson 1997) is suggesting that number of juveniles that reaches one year of age could be as high as 326 individuals for every 100 million eggs (0.000326 % survival rate) if trochus are fast growing, reaching sexual maturity one year after settlement. Survival rate could be much higher at 1 230 individuals for every 100 million eggs (0.00123 % survival rate) if the trochus are slow growing and can only attain sexual maturity in two years. Because reproduction is size dependent (upon reaching a size of > 50 mm irrespective of age, see Nash 1993), this implies that faster population growth rate is likely to occur in areas that promote faster growth as they may tend to reproduce at early age, than in areas where growth is stunted and reproduction may occur at later age. In this case, conservation actions that promote the survival of trochus juvenile in areas where trochus can have faster growth are likely to lead to a successful outcome.

Predation and many other factors could have a high impact on eggs, larvae and juveniles after settlement (Andrew and Choat 1982; Watanabe 1984; Yamada and Boulding 1996; McArthur 1998; Ray-Culp et al. 1999; Beal 2006). The survival of wild trochus from eggs until settlement is not known but for many fish species, the survival of eggs can be very low and can vary depending on mode of reproduction and many environmental factors (Dahlberg 1979; Leggett and Deblois 1994). Egg survival is generally higher (can range between 16 – 95 % depending on species) for species that exhibits parental care than for species which do not guard their eggs (< 1 to 25 % depending on species). The survival rates from eggs up to larval stage for freshwater Northern Pike (0.03 % survival after 45 days) and Atlantic herring (0.000004 % survival after 77 days) (see Dahlberg 1979) suggests that very few individuals would reach sexual maturity. Among newly settled barnacles, low survival rates on the first day of settlement may constitute a bottleneck for survival as these were neither related with densities of recruits, abundance of grazers or desiccation stress and wave exposure (Gosselin and Qian 1996). Trochus are highly fecund broadcast spawning gastropod (Nash 1993; **Chapter 4**), they do not exhibit parental care, thus only a small fraction of the spawned eggs would reach sexual maturity.

The vulnerability of juveniles to predators could depend on their size, nature of habitat and presence of conspecific adults. Compiled studies on juveniles of various invertebrate species indicates that mortality is generally higher at early stages (Gosselin and Qian 1997) because smaller sized prey are generally highly vulnerable to predators (Moran 1985). The juvenile trochid gastropods *Tegula* spp. find refuge on dense vegetation, but predation pressure increases with depth due to decreasing vegetation cover and more abundant predators in deeper waters (Watanabe 1984). The presence of conspecific adults has contrasting effects on the survival of juvenile invertebrates. For example, the probability of juvenile survival of abalone *Haliotis rubra* decreases as adult density increases (McShane 1991). For the echinoid *Evechinus chloroticus* (Andrew and Choat 1982) and the bivalve *Macoma balthica* (Richards et al. 2002) caging experiments did not show any effect of the density of adults on numbers of juveniles. By contrast, the survival of juveniles of queen conch *Strombus gigas* (Ray and Stoner 1995; Ray-Culp et al. 1999) and bryozoans (Keough 1986) are positively associated with adult density, and the presence of adults suggests that it is a good place to live. For the case of trochus, juveniles are only encountered at the intertidal area, often in groups of 2 – 4 ind m^{-2}, and their size increases with size of rocks (Castell 1997). Trochus occur in higher densities at unexploited intertidal habitats than in subtidal areas (Nash 1993; **Chapter 2**), which suggests a higher dependence of settling juveniles on adult density at intertidal than at subtidal areas, or it could reflect migration or higher mortality subtidally. However density dependence can reduce juvenile survival and slow down population growth when space and resources are becoming limited. The population size increase in sea otters (*Enhydra lutris*) is negatively related to population growth rate (Gerber et al. 2004a). Experiment on elevating the density of adult temperate reef fish cunner *Tautogolabrus adspersus* to "twice the natural levels have resulted to a complete mortality of newly settled fish, while removal of adults resulted in enhanced growth and success over natural populations" (Tupper and Boutilier 1995).

The elasticity analyses on the age-structured model for both slow and fast growing unexploited trochus revealed that survival of the sub- and young adults had the greatest elasticity value. More than 50 % of survival elasticity values fall within the age classes 1 – 3 years, which suggests their higher contribution to population growth compared to other age classes. Therefore there is a need to focus the conservation action among sub- and young adults over the protection of very large individuals or the mass production and release of juveniles "headstarting". The gradual increase in survival elasticity for the smaller age group and gradual decline for older age group with the reduction in P_i (from 90 to 75 %) in unexploited trochus population suggests the increasing contribution of younger age classes to population growth with declining adult survival rates. This implies again that in reviving an overly exploited trochus population, conservation action should focus on the promotion of survival of sub- and young adults.

A pattern whereby survival elasticities are larger than fecundity elasticities has been reported for many species such as abalone, marine turtles, sea otters and species of corals (Gerber et al. 2004b; Rogers-Bennett and Leaf 2006 and references therein). Long lived and slow growing species are expected to have a high juvenile or adult survival elasticities and low fertility elasticities (Heppell et al. 2000; Rogers-Bennett and Leaf 2006) because increasing the survival produces more eggs for the rest of individual's life. Benton and Grant (1999) also confirmed that in general, survival elasticities are greater than fecundity elasticities, but with declining degree of differences as generation time decreases.

Under extreme fishing condition ($P_{1\text{-}12}$ = 10 %), the elasticity values are confined to smaller age classes and later age classes have very low elasticities because very few old or large individuals remained in the population as a result of natural and fishing mortalities. The higher fecundity elasticities compared with survival elasticities for heavily fished slow and fast growing trochus suggest that allowing the individuals to breed is the only option to revive their population.

Several conservation measures which firstly focused on "headstarting" have shifted to strategies that promoted the survival of adults when elasticity analyses have shown that large juveniles and adults have higher survival elasticities. The case of endangered marine turtles such as the loggerhead sea turtles *Carreta carreta* and Kemp's ridley sea turtles *Lepidochelys kempi* and the non-threatened yellow mud turtles *Kinosternon flavescens*, are classic examples where the protection of nesting sites and rearing of hatchling for several months "headstarting" as a conservation efforts was conducted for 15 years before elasticity analysis demonstrated that adult survival had a greater effect on population growth. Such findings led to the development of turtle excluding devices on fishing gears to promote the survival of large turtles (Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1996). The biggest elasticities on trochus are for adult survival, similar to that of turtles and other long lived repeat breeding mammals which is not what we expect from a snail, therefore, to conserve the species, it is important to promote the survival of these adults and allow them to breed.

Under favourable conditions, trochus growth can be fast, reaching sexual maturity in one year although it has been reported that on the average, trochus become sexually mature upon reaching a size > 50 mm at the age of two years (Nash 1993). Heslinga (1981) has shown that trochus in Palau can grow faster and reach sexual maturity in one year under favourable laboratory conditions. Trochus in the wild in mainland Palawan have the same growth rate (**Chapter 6)** as the report of Heslinga (1981), while captive 3-year old trochus were successfully bred upon reaching a size of > 50 mm (**Chapter 3**). Presence of tubercles and spiral ridges was also reported for a large (63.5 mm) single wild specimen from Aitutaki, Cook Island a few years after the transplantation in 1957.

The tubercles on trochus shell usually disappear at size 25 – 30 mm (Purcell 2002), that the presence of tubercles for trochus with 63.5 mm shell diameter was attributed to fast growth (Devambez 1960). This suggests that under favourable conditions, trochus in the wild can reach a sexual maturity size of > 50 mm in one year.

The conservation of trochus through the protection of adult individuals is not new. A number of countries have been implementing size limits along with a combination of other conservation measures (Nash 1993; Purcell 2004) to allow trochus to breed for at least 1 – 2 years before harvesting (Nash 1993). The minimum size limit varies (64 – 96 mm) between countries (**Figure 7.7, Table 7.4**) because factors such as growth rate, size at maturity and maximum attainable length differ between countries (Nash 1993). The use of minimum and maximum size limit is a more conservative measure than minimum size limits alone, because it restricts the capture of undersized shells and very large highly fecund adults. The use of size limit however works well when fishing pressure is moderate which allow some individuals to breed and grow larger, or should be combined with catch quota to achieve some large reproducing trochus in the population (Purcell 2004).

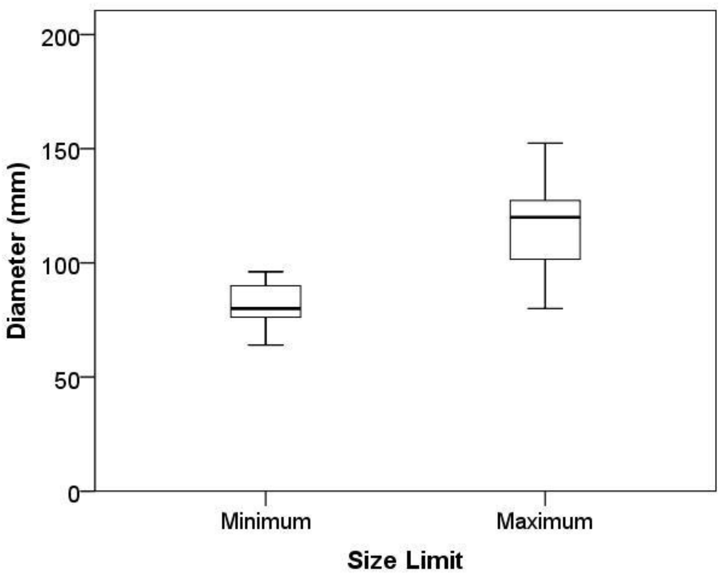

Figure 7.7. Minimum and maximum size limits (mm) for trochus fisheries management set by different countries in the Indo-Pacific Region.

Table 7.4. Minimum and maximum legal size limits (mm) set in harvesting trochus in different countries.

Country	Min. Size Limit (mm)	Max. Size Limit (mm)	Sources
Solomon Islands	63.5	80.0	(Amos 1997)
	80.0	120.0	(Lasi 2010)
Western Australia	65.0	100.0	(Magro 1997)
Chuuk	76.2	152.4	(Amos 1997)
Kosrae	76.2		(Amos 1997)
Pohnpei	76.2	101.6	(Amos 1997)
Yap, Micronesia	76.2	101.6	(Amos 1997; Fanafal 1997)
Palau	76.2		(Heslinga et al. 1984; Amos 1997)
Cook Islands	80.0	110.0	(Amos 1997; Tuara 1997)
French Polynesia	80.0	110.0	(Amos 1997; Cheneson 1997)
Papua New Guinea	80.0	130.0	(Anonymous 1997)
Queensland, Australia	80.0	125.0	(Nash 1988; Gillespie 1997)
Fiji	89.0		(Amos 1997; Ledua et al. 1997)
New Caledonia	90.0	120.0	(Amos 1997)
Vanuatu	90.0		(Amos 1997)
India	90.0		(Krishnamurthy and Soundararajan 1999)
Okinawa, Japan	60.0		(Isa et al. 1997b)
Cooperatives in Japan	Mean (±sd) of 111.7 ± 15.6 mm and 112.7 ± 17.5mm		(Isa et al. 1997b)

Elasticity analyses have shown that managing the stock requires a larger size limit (at least 98 mm), but none of the trochus producing countries implement a minimum size limit this big. Two cooperatives in Japan however have possibly imposed a higher size limit because their average catch is > 110 mm in diameter. The highest minimum size limit (89 – 90 mm) was implemented only in Fiji, New Caledonia, Vanuatu and India, while many other countries allowed the harvest of 64 – 80 mm individuals, the sizes having the highest survival elasticities in the model (**Table 7.4**).

The conservation of trochus by size limit is always faced with problems related to harvesting of sizes below or above the imposed size limits. In Solomon Islands, fisheries regulations prohibit the harvesting of trochus smaller than

80 mm and larger than 120 mm basal diameter, however, a sharp fall in trochus production since the mid 1990s indicates that this did not control overfishing (Lasi 2010). Without effective monitoring, undersized trochus often form part of fisher's harvest and this would only exacerbate population decline if the juveniles are being harvested. In Saipan, the minimum size limit is 76 mm but trochus shells confiscated by government authorities from a shipping company in 1996 contained about 38 % of undersized shells (Trianni 2002). In New Ireland, Papua New Guinea, 22 % of trochus exported in 1989 were below the 80 mm minimum size limit and about one percent was larger than the 130 mm maximum size limit (Anonymous 1997). When quota or close season is implemented, fishermen might also harvest the shells during the close season, and sell them only during the open season.

The harvesting of undersized adult trochus could have a big impact on its population growth because elasticity analyses have shown that heavily exploited population requires a large number of surviving juveniles to maintain a stable population growth ($\lambda = 1$). When the survival of the first 3 age classes was held constant at either 75, 82 or 90 % and the survival of the remaining age classes were held constant at 10 %, juvenile survival for slow and fast growing trochus should range between 78 – 115 ind per 100 million eggs and 40 – 54 ind per 100 million eggs respectively so that $\lambda = 1$. By contrast, a much higher juvenile survival is required when the first 3 age classes were maintained at 10 % survival rate and the survival of rest of the older individuals were held constant at either at 75, 82 and 90 %. The values of the juvenile survival for slow and fast growing trochus should range between 2 118 – 2 249 ind per 100 million eggs and 210 – 211 ind per 100 million eggs respectively, and are comparable to the required juvenile survival under heavy fishing condition (10 % survival for all age group).

Effective law enforcement and increasing fishermen's understanding of the consequences of overharvesting can help reduce the problems with undersize harvesting (Purcell 2004). Pollnac et al (2010) also found that the levels of compliance with marine reserves rules were related to complex social dynamics rather than simply enforcing.

Successfully managed trochus fisheries in Cook Islands and Palau was achieved through tightly controlled open and closed seasons in addition to imposing maximum and minimum size limits (Lasi 2010). In the Philippines, Gomez and Mingoa-Licuanan (2006) have emphasise the importance of community cooperation and involvement in their long-term project on giant clam restocking. In Vanuatu, trochus resources was successfully managed through conservation initiatives of the local stakeholders (Dumas et al. 2010).

The preceding approaches involving the protection of trochus that allowed them to repeatedly reproduce over time were found to effectively revive overharvested population. These corroborate with the results of elasticity analyses that trochus allowed to breed in the wild is the only option to successfully manage the remaining wild population. Trochus is long-lived large reef invertebrate and therefore a successful management strategy for this species could be applied to similar large long-lived repeat-breeding reef invertebrates. However, the juvenile survival rates obtained in this study were estimated from age-structured model and secondary information on growth of newly established populations. Empirical data on fertilisation rate, juvenile survival and density dependence in the wild of trochus or of any studied species can help refine the estimates in the model. As growth and survival of trochus may be site specific, prior knowledge on these parameters is important when applying the model to a particular site.

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Chapter 8

CONCLUDING REMARKS

The trochus fishery underwent a boom at the start of its exploitation in the early 1900s, and rapidly became overharvested. Several studies have reported the decline of trochus fisheries immediately after the start of commercial harvests (Nash 1993; Etaix-Bonnin and Fao 1997; Hahn 2000; Bell et al. 2005). This thesis has shown how exploitation can massively reduce the abundance of trochus in protected sites in a very short time, changing their pattern of distribution so that they become restricted to the deeper parts of the reef. Nevertheless, recruitment continues even in heavily exploited MPAs in Palawan on the mainland (**Chapter 2**). Non-conventional methods to conserve trochus like sea ranching met with limited success (Crowe et al. 1997; Isa et al. 1997; Crowe et al. 2002; Purcell et al. 2004) and this was also the case in Palawan, Philippines. Efforts to enhance trochus population by sea ranching pioneered by a private corporation have successfully bred and produced trochus juveniles and released millions of juveniles and larvae in nearly 10 years, but there was no harvest at all, forcing the company to close down (**Chapter 3**). Trochus are long-lived animals (Smith 1987; Nash 1993; Lemouellic and Chauvet 2008) and can breed at least 3 times in two years, with a portion of population reproducing every month (Hahn 1993), but the method of rearing wild adult trochus in indoor tanks after induction of spawning had resulted in high mortality rates, halting their potential reproductive activity on the reef (**Chapter 3**). In **Chapter 4** we demonstrated that keeping of wild trochus broodstock in intertidal tanks can increase their survival rates and allowed them to spawn naturally, possibly with a much higher fertilisation rate than when scattered in the wild. Trochus kept in this condition were successfully induced to spawn after nearly a year in captivity. The rearing of high numbers of trochus juveniles in indoor tanks prior to release is often beset with problems of space, food shortage, high mortality (Nash 1989; Nash 1993; Isa et al. 1997) and requires intensive labour (Amos 1997). Various methods of intermediate culture using sea cages have been tested with varying degree of success (Purcell 2001;

Amos and Purcell 2003). In our intermediate culture of trochus in indoor tanks and sea cages, trochus have high survival rates (nearly 100 %) in four growth experiments held either indoor or in cages on the reef slope (**Chapter 5**). In one of the growth trials on cages set on the deeper part of the reef, a high growth rate of > 5 mm mo^{-1} was attained (as high as in the wild) during the first 60 days of rearing at a density of about 100 ind m^{-2}. In **Chapter 6**, we have found that the translocated wild juvenile and adult trochus have high survival rates with growth rates that were either faster or slower (possibly due to food abundance and effect of shifting sand) relative to the growth of non-translocated trochus occurring at high densities. In **Chapter 7,** elasticity analyses on the age-structured matrix model showed that sub- and young adult trochus have higher survival elasticities than fecundity elasticities suggesting that increasing the survival of this age group would have a higher impact on population growth.

This thesis has some limitations which mean that its application to other species or location should be done with caution. (1) Our abundance survey was limited to a depth reachable by breath-hold dive and so does not provide information on the state of trochus in deeper parts of the reef. However, as fishermen from the southern Palawan are venturing in Malaysia (pers. comm. with residents from the southern Palawan) in hope for a good harvest, this suggests that trochus in Palawan occurred in low numbers. Our abundance survey in heavily exploited MPAs at the mainland Palawan have shown continued recruitment, but this may not be the case for all other reefs around the country. Many MPAs have been protected for several decades but most researches focused only on successful fish recovery (Russ and Alcala 1996; Alcala 1998; Russ and Alcala 2003; Russ et al. 2004; Abesamis and Russ 2005; White et al. 2005). Very little work has been done on invertebrates, but that which has long generally involved the release of mass-produced juveniles (Maliao et al. 2004; Gomez and Mingoa-Licuanan 2006; Lebata-Ramos et al. 2010) and suggested a limited recovery (Raymundo 2003). Trochus recovery on MPAs has only been examined at TRNP (Dolorosa et al. 2010). Therefore, abundance surveys are needed at each site to serve as guide to propose the best conservation option for trochus and other species of interest.

(2) Although we were able to successfully increase the survival of broodstock and able to induce them to spawn, the intertidal tanks used to hold trochus breeders are frequently exposed to extreme temperatures in summer day low tides, which we minimised by putting a shade above each tank. Extreme temperature, salinity and food shortage were reported to affect the growth of trochus juveniles under laboratory condition (Gimin and Lee 1997; Yi and Lee 1997) and these extreme conditions in summer time might have an effect on the life span and reproductive output of adult trochus kept at intertidal tanks. Splight and Emlen (1976) found that spawning frequency of the gastropod *Thais emarginata* was closely correlated with food supply. (3) The use of subtidal cages with coconut leaves as substrate can promote high survival and growth rate, but the construction of cages means additional cost and the setting up on the deeper part of the reef require more labour than at shallow protected sites. However, a site less affected by waves might have a limited amount of setting organic particles that adding of seaweeds as food may be needed to enhance the growth of trochus. In addition, improper site selection and lack of care during deployment of cages can cause damage to coral reefs. The high growth rate of trochus juveniles in cages is not a guarantee that the trochus will survive once released in the wild. Sub-adult hatchery produced trochus are still prone to predation (Villanueva et al. 2010) and acclimation of trochus to potential predators before releasing might increase chances of survival. Acclimation to natural habitats and natural predators of hatchery produced species prior to release were found to increase survivorship (Davis et al. 2005; Kawabata et al. 2011). (4) The reef where translocation is intended needs to be properly assessed. Trochus have a patchy distribution at TRNP which suggests that some environmental parameters are affecting their distribution. Suitable areas bordered by sand do not support trochus populations (Colquhoun 2001). There were a number of unsuccessful translocations of wild trochus (Gillett 1989; Gillett 1993) possibly due to improper habitat selection. Areas known to have had high abundance of trochus in the past could be good sites as long as the translocated trochus are protected from harvesting. Long-term monitoring of sites of re-introduction of wild trochus and monitoring of control sites (without re-introduction) is needed to find out whether the reintroduced

trochus have ultimately established. Cost benefit analysis is also needed to provide a more conclusive evidence of the benefits derived from translocation.
(5) In modelling the response of different age groups of trochus to perturbation, secondary information on fecundity was used but this has proven useful in predicting the age classes with the highest elasticities. It is generally normal for long lived species to have a higher survival than fecundity elasticities (Benton and Grant 1999; Heppell et al. 2000; Rogers-Bennett and Leaf 2006) but variation may occur even for species of the same genus. For example, the largest size classes of white abalone *Haliotis sorenseni* are more sensitive to perturbation, while it is the sub- and young adults of red abalone *H. rufescens* (Rogers-Bennett and Leaf 2006). This means that the sizes which require the focus of conservation differs between the two species, hence elasticity analysis is an important step before engaging in any wide scale conservation measure of a particular species of interest.

Implications for conservation

Coral reefs and their associated fauna are globally important in terms of fisheries, coastal protection, tourism, scientific studies, aesthetic value and many other goods and services (Moberg and Folke 1999; Ablan et al. 2004; Wilkinson et al. 2006). However, coral reefs are not well managed and their resources are in serious decline (Hodgson 1999; Wilkinson 2000; Hodgson and Liebeler 2002). In the Philippines, coral reefs are heavily degraded due to human activities (Gomez et al. 1981; White and Cruz-Trinidad 1998; Raymundo et al. 2007) and unregulated harvest of reef associated fauna (bin Othman et al. 2010; Hodgson 1999; Raymundo 2003; Ablan et al. 2004). Aside from these direct human actions, coral reefs are severely affected by climate change, causing wide scale bleaching (Obura 2005; Crabbe 2008; Schleyer et al. 2008). Well-managed reefs having high populations of herbivores may be more resilient to impacts of climate change (Baker et al. 2008).

Overharvesting of herbivorous reef-associated fishes can cause a shift from corals to macro-algal dominated ecosystems. Herbivores can control macro-algal blooms (Jones 1992; Ceccarelli et al. 2011) brought about by the increasing influx of nutrients from agriculture and domestic wastes. The dominance of perennial macro algae and reduction in essential reef components indicates a degraded reef (Bahartan et al. 2010). The protection of reef ecosystems can increase biodiversity and sustain marine capture fisheries (Russ et al. 2004; Abesamis and Russ 2005; Russ and Alcala 2011). As sea ranching has failed to provide evidence of success in enhancing trochus populations (Crowe et al. 1997; Isa et al. 1997; Crowe et al. 2002), restoring trochus populations in the Philippines may therefore require a different approach than the current headstarting method. Conventional methods involving the active participation of community have proven effective (Pollnac et al. 2001; Pollnac et al. 2010) in resource management, which was also proven effective in reviving trochus populations (Dumas et al. 2010).

The trochus population in the Philippines needs to be revived where it is an important fishery commodity. The presence of recruits in heavily exploited sites provides evidence that trochus are resilient to exploitation and respond well when their habitats are protected. The depth that fishermen can exploit has been limited following the prohibition of hookah fishing and this has possibly allowed the trochus in deeper parts of the reefs to reproduce. These areas were not surveyed during this study but contracted fishermen have collected trochus by breath hold dive from deeper parts of the reef. Surveys of these habitats involving the use of underwater breathing apparatus could provide information on the population size and structure of trochus in the unexploited parts of the reef around the mainland Palawan, which could in turn be used to predict population growth when the populations are protected.

The high survival rates of wild translocated juveniles suggest the possible use of recruits for translocation. These could be easier to handle or transport than large individuals. In our perturbation analyses on the age-structured matrix model for trochus, sub- and young adults are highly sensitive to perturbation suggesting that these age groups have higher contribution to population growth. Thus for conservation efforts to be effective, focus should be shifted in promoting the survival of this age group in the wild. As government find it hard to effectively protect large MPAs, translocation into well-protected suitable areas known to hold trochus in the past but is not showing recruitment could be the best alternative. Well-managed village MPAs (Dumas et al. 2010) or privately-managed reserves (Svensson et al. 2008 ; 2010) is hoped to revive trochus population.

Elasticity analyses found that the survival of age classes of slow (1-3 years old or < 73 mm) and fast (1-3 years old or <100 mm) growing adult trochus have the highest contribution to population growth, thus these sizes needs to be protected. However, imposing a multiple size limit on the part of government could be difficult to monitor that it is rather practical to set a higher size limit which would protect all the population irrespective of their growth rates. Setting a higher size limit should also be carried out along with effective support to the villages with respect to education information, alternative livelihood while the reef is closed for fishing, and active involvement of the community in conservation measures.

Future works and challenges

Reviving trochus back to its state prior to the start of commercial exploitation is an enormous challenge to government and conservation workers, but the prohibition of cyanide fishing (often associated with hookah fishing) under the Philippine Republic Act 8550 otherwise known as the Philippine Fisheries Code of 1998 (DA 1998) and prohibition of hookah or compressor fishing under the local government units (e.g. City Ordinance No. 267 of Puerto Princesa of 2004) could be the reason for the continued recruits observed in heavily exploited sites. However, the challenge now is on how to control the continued gleaning at reef flats which form part of MPAs. Continued collection of sub-adult trochus or even continuous trampling on their habitats can hold back population growth. A number of studies have shown that trampling alone can have a negative effect on intertidal biodiversity (Brown and Taylor 1999; Milazzo et al. 2004; Casu et al. 2006).

Although trochus have been declared a threatened species (DA 1998 ; 2001; Floren 2003), harvesting continues illegally and many trochus of all sizes are collected for subsistence use. This explains why trochus populations were not improving in surveyed MPAs. With the decline of trochus density at TRNP due to poaching (Dolorosa et al. 2010; Jontilla et al. in press), and the continued harvest of trochus in all parts of the country (Del Norte-Campos et al. 2003; pers. comm. with residents from southern Palawan), the effective Philippine law enforcing system is urgently needed along with long term monitoring of areas subjected to protection and areas without management interventions for scientific comparison. It is also important for any conservation measure to include the derived economic benefits to provide a stronger claim on its effectiveness and to strengthen conservation advocacy among the stakeholders. Cost-benefit analyses in the reseeding of hatchery produced abalone (Schiel 1993) and for a number stock enhancement programs in Japan (see review of Masuda and Tsukamoto 1998) have been reported.

The variation in distribution of trochus at deeper parts of the reef is an interesting topic to investigate. The abundance of trochus at 5 m on the deep reef slope at TRNP was very low at < 20 ind ha^{-1} (Jontilla 2008), although large specimen can occur at up to about 20 m deep where they are usually covered with crustose algae (Nash 1993; Smith et al. 2002). Surveys of Galon et al. (2007) of 4 – 21 m deep reef inside an MPAs in the mainland Palawan revealed a density of 190 ind ha^{-1} while no trochus were recorded at an unprotected site. The tendency of trochus to occur at shallow reefs at TRNP makes them vulnerable to over-exploitation. Trochus density at permanent monitoring sites at TRNP was reduced by about 70 % from a mean density of about 6 000 ind ha^{-1} in 2006 (Dolorosa et al. 2010) to nearly 2 000 ind ha^{-1} in 2008 (Jontilla et al. 2011) presumably by poaching. Finding out the factors affecting the vertical distribution of trochus at TRNP could shed more light in understanding the ecology of this highly sought after reef gastropod.

The breeding of trochus can be carried out successfully, but more studies are needed to increase the survival of released juveniles in the wild. The intermediate culture of trochus in deep subtidal cages using light plant materials as substrate can help reduce the cost of hatchery production, increase juvenile survival and hasten growth. However, setting of cages at deeper reefs requires additional cost and labour. Too many cages set on the reef might also hinder the growth of corals and other marine life. When typhoons come, cages can cause damage to corals when they are blown by waves into the shore. Setting the cages on rock substrate at shallow semi protected sites could be another option. Another potential option for intermediate rearing of juveniles is by using the intertidal tanks provided with cover which allow only the entry of small predators. In this case, the trochus juveniles would be more acquainted with the danger of predation which can help increase their survival in the wild.

Harvesting of trochus continues even when it is declared a protected species (Dolorosa et al. 2010). The reef flat which form part of studied MPAs at the mainland Palawan were frequented by gleaners and all sizes of trochus were harvested, even those released from the hatchery. This practice of harvesting the recruits before reaching the sub adult size can further increase the vulnerability of trochus to extinction even when access to deeper parts of the reef is limited due to the prohibition on the use of hookah breathing apparatus. Trochus are more visible on the reef when they breed (Nash 1993) and local fishermen at the mainland Palawan are aware of time or phase of moon or tide condition where trochus are more visible on the reef. In such a case, there is no hope for trochus to recover its population even when it is declared a threatened species. Since fishing and gleaning activities at the intertidal areas of village MPAs continues, this activity can inhibit the trochus population growth. Trochus are relatively easy to harvest on the reef at the mainland Palawan that are accessible to fishermen with small outrigger canoes and diving goggles. Collected trochus are easy to hide and sell which could also be attributed to lax law enforcement. Only in TRNP where arrests for trochus collection have been reported since 2007 (Dolorosa et al. 2010; Jontilla et al. 2011). Between 2007 and 2011, there were 15 apprehensions involving 183 local fishermen. About 55 cases have been filed and 37 fishermen were convicted (TMO unpublished data).

Lastly, I would like to quote the abstract from the paper of Sale (2008): "Globally, our current management of coral reefs is inadequate and becoming more so as we place new and greater stresses on these ecosystems. The future looks very dim, and yet we have the capacity to do a far more effective job of reef management if we want to. Making substantial improvements to the condition of these enormously valuable coastal marine ecosystems does not require new scientific discoveries, but a new commitment to apply the knowledge we already possess to manage our impacts so that sustainability becomes possible". Therefore, strong commitment of involved government and stakeholders entities is needed to successfully implement any of the known conservation measures. Long term monitoring of areas with and without interventions are needed to provide vital information on how the system works under a set of given circumstances, and on how to improve the existing management actions through time.

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