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Patterns and mechanisms of spillover from marine reserves

Thesis submitted by
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within the School of Marine Biology and Aquaculture
James Cook University

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Rene A. Abesamis

Abstract

The use of no-take marine reserves (areas closed to fishing) as fishery management tools has been advocated considerably in the past decade. No-take reserves are now often established with an objective of enhancing fisheries. However, the fishery effects of no-take reserves remain controversial. Spillover, or net export of adult biomass, is one potential process of fisheries enhancement by reserves. Few studies have shown convincingly that reserves can develop spillover that benefits local fisheries. The objective of this thesis was to investigate the predicted patterns and proposed mechanisms of spillover from no-take reserves. The no-take reserves studied were on coral reefs in the Philippines that have been well protected for 15 to 20 years.

In the first study, I tested the prediction that gradients of decreasing abundance of target species, from high inside the reserve to low outside, would be present across the boundaries of no-take reserves. Spillover is predicted to produce such patterns. Underwater visual census of fishes was carried out along large transects (each one approximately 1 km long) that crossed the lateral boundaries of each of two no-take reserves and two controls (fished sites) at each of two islands. Patterns of decreasing abundance (density and biomass) of target fishes were found only across one boundary of one no-take reserve, the reserve at Apo Island (Apo Reserve). The patterns of decreasing abundance were limited to within 50-200 m of the boundary. Rates of decrease in abundance across the boundary were two to three times greater for sedentary fishes than for vagile fish. This finding is consistent with the expectation that the spatial extent of spillover would vary according to mobility of fishes. The patterns of decreasing abundance of target fishes were probably not due to gradients in habitat. Furthermore, no pattern of decreasing abundance across the boundary was found for fishes not targeted by fisheries.

In the second study, I investigated the potential mechanisms inducing movement of a targeted planktivorous reef fish, *Naso vlamingii*, from Apo Reserve to adjacent fished areas. Repeated underwater visual census at reserve and fished sites suggested that some adults of *N. vlamingii* could emigrate temporarily from Apo Reserve within the short-term (days). Data from visual census and experimental hook and line fishing suggested that short-term emigration of *N. vlamingii* is related to movement to feeding areas 150-200 m away from reserve boundaries. However, some larger (41-45 cm TL) adults probably tend to stay inside the reserve. Experimental hook and line fishing adjacent to the reserve also showed that the mean size of *N. vlamingii* captured decreased as one moved up to 300 m away from the boundaries of Apo Reserve. This pattern may have resulted from density-dependent home range relocation of smaller adult *N. vlamingii* over the long term. Competitive interactions between adult *N. vlamingii* were up to twice as frequent, on a per fish basis, inside the reserve than outside. When interacting adults differed in sizes, the larger adult was always observed to chase away the smaller one. The sizes of adults that were chased away (25-35 cm TL) were similar to the sizes of those caught by experimental fishing (26-38 cm TL).

In the third study, I attempted to quantify the contribution of spillover to the yield of the local fishery at Apo Island. Daily fishing effort and yield (biomass and value) of local fishers in relation to Apo Reserve were studied for eight months (July 2003 to February 2004) over two monsoonal seasons. Average catch rates and income rates of fishers were found to be highest near the reserve (within 300 m of reserve boundaries). Catch rates near the reserve were higher than in other areas far from the reserve by a factor of 1.1 to 2.0, depending on the fishing gear. Moreover, the highest average monthly catch rates were always found near the reserve. These findings are consistent with occasional very large catches near the reserve due to spillover. However, fishing effort was often lowest near Apo Reserve. Furthermore, local fishers seemed to have avoided fishing very close to the reserve (within 100 m of reserve boundaries). Most fishing (79% of overall fishing effort) occurred far from the reserve, on the northern fishing grounds of Apo Island. Yields from spillover probably contributed less than 10% of the total yield of the local fishery. Fishing effort near Apo Reserve appears to be limited by 1) weather conditions caused by the monsoons, 2) the traditional importance of the northern fishing grounds, 3) high variability of catch rates and income rates, 4) lower value of target species found near the reserve, and 5) social pressures within the local community.

The results of this thesis provide consistent, yet equivocal, evidence for spillover from Apo Reserve. This study could not provide information on patterns of abundance (density, biomass, mean size or catch rate) and behaviour (movement and aggressive interactions) of target fishes, nor patterns of fishing effort of fishers, before the reserve was established. Nonetheless, research over the last two decades suggests that spillover of target fishes from Apo Reserve has developed over time. Spillover may have produced some of the patterns found in the present study. The main conclusion from this thesis is that Apo Reserve has probably developed spillover, but has provided very limited direct benefits to the local fishery.

Acknowledgements

This thesis would not have been possible without the excellent supervision of Garry R. Russ. I am also grateful to Angel C. Alcala for valuable advice and inspiration. Russ and Alcala's laudable work on Philippine marine reserves over more than two decades provided the foundation for this thesis. Figures 3.6 and 3.7 are based on Garry's data from Apo Island collected starting in 1983. I was then only seven years old.

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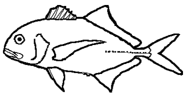

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Statement on Sources

Declaration

I declare that this thesis is my own work and has not been submitted in any form for another degree or diploma at any university or other institution of tertiary education. Information derived from the published or unpublished work of others has been acknowledged in the text and a list of references is given.

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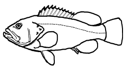
Rene A. Abesamis



Chapter 1

General Introduction

The terms ‘marine reserves’ and ‘no-take marine reserves’ are used in this thesis to simply mean marine areas where fishing is prohibited permanently. Marine reserves have been promoted extensively as fishery management tools in the past decade (Plan Development Team 1990, Roberts & Polunin 1993, Bohnsack 1996, Dayton et al. 2000, Russ 2002). Exploited species are predicted to build up in density, biomass and reproductive output inside no-take marine reserves over time due to the reduction or elimination of fishing mortality (Roberts & Polunin 1991, Rowley 1994, Bohnsack 1996, Russ 2002). These potential effects inside reserves form the basis for arguing that reserves could provide insurance against overfishing (Plan Development Team 1990, Pauly et al. 2002, Russ 2002) by becoming net exporters of target species biomass. This view has become more prominent in recent years, since conventional fisheries management (e.g. controlling catch or fishing effort) has been unsuccessful in preventing overfishing worldwide (Roberts 1997, Pauly et al. 2002).

In the developing world, reserves are viewed as one of the few viable approaches in managing fisheries on coral reefs (Alcala & Russ 1990, Roberts & Polunin 1993). Reef fisheries in developing countries are typically multispecific and involve different fishing gears (Russ 1991, Munro 1996). This makes conventional fisheries management expensive and time-consuming (Russ 1991). Developing countries simply do not have the resources to implement conventional fisheries management approaches (Johannes 1998). Management of reef fisheries is urgently needed in many areas of these countries. No-take reserves offer ecosystem-based fisheries management that could be implemented with less need for resources (Roberts & Polunin 1993, Bohnsack 1996). Furthermore, reef fisheries in developing countries support people who are among the poorest members of society. Very few alternative sources of income are available for these people. Implementing some of the standard fisheries management measures (e.g. size or bag limits, seasonal closures) in the developing world would be futile or even socially immoral.

A major objective of establishing no-take reserves is to enhance fisheries. To achieve this, reserves must eventually display net biomass export (Russ 2002). Reserves should also compensate for the initial loss in fishery yield that would result from the decrease in fishing area after their creation. Net biomass export from reserves may occur in two ways (Russ 2002). Firstly, reserves may become net exporters of larvae, if protection from fishing results in increased spawning stock biomass inside reserves. This mechanism is usually referred to as 'the recruitment effect'. Secondly, reserves may become net exporters of mobile adults, if protection from fishing results in higher adult density and biomass inside reserves. This mechanism is termed 'the spillover effect', or simply 'spillover'. This potential spillover effect of reserves on fisheries is the subject of this thesis. The focus of the thesis is spillover of fishes from no-take marine reserves on coral reefs.

It is important to stress from the outset that net larval export is probably the more important process of fisheries enhancement by reserves. There seems to be general agreement on this (Carr & Reed 1993, Sladek-Nowlis & Roberts 1999, Russ 2002). Fishery gains from spillover are predicted to be minimal (Polacheck 1990, Russ et al. 1992, DeMartini 1993, Attwood & Bennett 1995). Furthermore, spillover will probably be more localised than net larval export (Roberts & Polunin 1991, Russ 2002). However, demonstrating spillover may be critical in encouraging establishment and maintenance of reserves by local fishing communities (Russ & Alcala 1996). Reserves require support from local fishers (Russ & Alcala 1999, Galal et al. 2002, White et al. 2002). Fishers are often sceptical of the benefits of reserves. Clear evidence for local fishery gains may be needed to convince fishers to support reserves. Support from fishers may be difficult to obtain on the pretext of enhanced fisheries through increased recruitment. Net larval export from reserves may seem ambiguous to local fishers, as the effects will likely manifest at broad spatial scales (tens to hundreds of kilometres). In contrast, large catches of adult fish near reserves could be appreciated directly by fishers. In other words, benefits from spillover may have a considerable positive impact on the attitudes of fishers towards reserves.

The fishery effects of reserves remain controversial. Little evidence exists for net larval export from reserves (Gell & Roberts 2002, 2003, Russ 2002). On the other hand, empirical evidence for spillover has increased in recent years (Gell & Roberts 2002, 2003). Here, I briefly review the evidence for spillover, with emphasis to studies on fishes and no-take reserves on coral reefs. Empirical studies related to spillover are of three types. Firstly, there are studies that have demonstrated higher abundance of target species nearer than further from the boundaries of reserves. Spillover is predicted to produce a gradual pattern of decreasing abundance of target species away from reserve boundaries (Rakitin & Kramer 1996, Kramer & Chapman 1999). Several studies have detected higher abundance (density, biomass and/or catch rates) of target fish near reserves on coral reefs (McClanahan & Kaunda-Arara 1996, Rakitin & Kramer 1996, Russ & Alcala 1996, McClanahan & Mangi 2000, Russ et al. 2003, 2004). These studies indicate that spillover of coral reef fishes will be limited to tens of meters to a few kilometres from reserve boundaries. However, some of these studies did not account for habitat potentially affecting the spatial pattern of abundance of target fish. Higher abundance of target fish outside but close to reserve boundaries may simply be due to habitat differences, rather than spillover. Nevertheless, a few studies are persuasive because development over time (several years) of a decreasing pattern of abundance of target fish away from reserves has been documented (Russ & Alcala 1996, Russ et al. 2003, 2004).

Secondly, there are studies that have demonstrated movement of fish across reserve boundaries (reviewed in Roberts & Polunin 1991, Russ 2002). Studies on movement of coral reef fishes suggest that spillover of fish from reserves on coral reefs would be localised for many species (Holland et al. 1993, 1996, Zeller & Russ 1998, Chapman & Kramer 2000, Meyer et al. 2000, Eristhee & Oxenford 2001). Furthermore, these studies indicate considerable potential for a mechanism of spillover involving occasional emigration of fish from reserves during daily or seasonal movements within home ranges (Roberts & Polunin 1991, Gell & Roberts 2002). It is also commonly assumed that even ‘random’ movement of fish across reserve boundaries may lead to spillover (Roberts & Polunin 1991, Russ 2002). Theoretical studies predict that random movement across reserve boundaries may result in a gradual pattern of decreasing abundance of fish from inside to outside reserves (Walters et al. 1999, Walters 2000). However, true random movement may not be prevalent in coral reef fishes (Holland et al. 1993, 1996, Zeller 1997, 1998, Zeller & Russ 1998, Meyer et al. 2000, Eristhee & Oxenford 2001).

Another hypothesised mechanism of spillover is net emigration of fish from reserves, due to density-dependent effects, such as increased competition inside reserves (Roberts & Polunin 1991, Kramer & Chapman 1999, Sanchez Lizaso et al. 2000, Russ 2002). Spillover caused by density-dependence should involve permanent relocation of home ranges, which may result in a pattern of decreasing density or mean size away from reserve boundaries over the long term (Rakitin & Kramer 1996, Kramer & Chapman 1999). Density-dependent movement of fish from reserves is a reasonable expectation, particularly for territorial coral reef fishes. However clear evidence for this mechanism of spillover is still not available. Only one study has attempted to examine experimentally the role of density on movement of fishes in relation to no-take reserves (Zeller et al. 2003). This study, however, did not detect significant changes in movement of fish across reserve boundaries, even after density gradients between reserve and fished sites were increased experimentally (Zeller et al. 2003).

Thirdly, there are studies that have monitored fisheries catch outside reserves. Several of these studies were made on coral reefs. One unique study demonstrated that catch rates and total yield of reef fishes decreased 18 months after a small reserve (Sumilon Reserve, Philippines; < 1 km^2), well protected for a decade, was opened to fishing (Alcala & Russ 1990). This study suggested that spillover contributed significantly to total yield of the local fishery for almost a decade. However, the magnitude of actual spillover yield was uncertain (Russ et al. 1992, DeMartini 1993). The remaining studies on coral reefs have monitored catch before and after creation of reserves (McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2000, Roberts et al. 2001, Galal et al. 2002, Maypa et al. 2002, Russ et al. 2004). All of these studies documented higher catch rates after reserves were established. However, in most of these studies the amount of yield generated by spillover was uncertain. Two studies suggested that spillover from a small reserve (Apo Reserve, Philippines; < 1 km^2) helped to maintain high annual yields of the local fishery for almost two decades (Maypa et al. 2002, Russ et al. 2004). However, no attempt was made to quantify actual spillover yields. Another two studies argued that total yield of fisheries improved five years after networks of reserves were established, since catch rates were higher but fishing effort remained constant [Roberts et al. 2001 (St. Lucia), Galal et al. 2002]. However, changes in total yield were not monitored, precluding estimation of the spillover contribution to total yield. Only two studies provided estimates of fishery yields from spillover (McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2000). From these studies, it can be deduced that spillover from a large reserve (Mombasa Marine Park, Kenya; 6-8 km^2) contributed 20-30% of total fisheries yield. However, spillover did not compensate fully for the loss in total yield that resulted from creation of the no-take reserve.

The magnitude of spillover, and its importance to local fisheries, will likely vary from case to case. Many variables will determine potential and actual spillover yields. Potential spillover yield (yield per recruit) will be a function of reserve size, fishing mortality outside reserves, and demographic parameters of target species, particularly population growth rate and movement rates of individuals (Polacheck 1990, Russ et al. 1992, DeMartini 1993, Jennings 2001). Recovery within reserves will also depend on initial population size (a function of historical fishing intensity), recruitment success, and degree of reduction of fishing inside reserves (Jennings 2001). On the other hand, actual spillover yield, and its value relative to total yield, will be determined by patterns in fishing effort in relation to reserves. It is commonly expected that fishers will begin to concentrate fishing effort near reserves in response to higher catch rates due to spillover (Walters et al. 1999, Gell & Roberts 2002). However, few studies have demonstrated higher fishing effort adjacent to reserves, or quantified catch rates and yields near reserves. Nevertheless, patterns of fishing effort may depend upon factors other than catch rates (reviewed in Hilborn & Walters 1992, p. 117). Patterns of fishing effort may be influenced by differential costs of travelling to fishing areas (e.g. Wilcox & Pomeroy 2003) and differential values of target species (Hilborn & Walters 1992). Furthermore, other factors not related directly to income (e.g. weather, local traditions and social pressures) may also affect patterns of fishing effort (Hilborn & Walters 1992, Wilcox & Pomeroy 2003). Successful use of reserves to enhance fisheries would require a case by case understanding of the spatial structure of impacted fisheries, ecosystems and human communities (Hilborn et al. 2004).

Three major points emerge from this review of the evidence for spillover:

1) The main pattern predicted for spillover is a higher abundance of target species nearer than further from the boundaries of no-take reserves. This pattern may be found in density, biomass, mean size and catch rates of target species. In addition, this pattern may be reflected in fishing effort, if fishers respond to spillover.

2) The hypothesised mechanisms of spillover are plausible but remain poorly understood. No study has demonstrated mechanisms of spillover from no-take reserve developing over time.

3) Few studies have quantified fishery yields from spillover. The magnitude of actual spillover yield, and its importance to local fisheries, will likely vary from case to case. Detailed studies of reserves and the fisheries external to them are required to improve our understanding of the fishery effects of reserves.

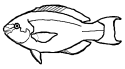
This thesis addressed these crucial points. The objective of the thesis was to investigate the predicted patterns and proposed mechanisms of spillover. In doing so, I used the following approaches: 1) underwater visual census, 2) experimental fishing, 3) behavioural studies, and 4) monitoring of fisheries. The studies that were made in this thesis were written up as three stand-alone chapters (Chapters 2-4) intended for publication.

In Chapter 2, I tested the prediction that gradients of decreasing abundance of target fish are present across the boundaries of well-protected no-take reserves. Intensive visual census of fish was carried out along large transects (each one approximately 1 km long) across each of two reserves (protected for 15 and 20 years) and two fished 'controls' in two islands in the Philippines. The effects of habitat on the patterns of abundance of fish were determined.

In Chapter 3, I investigated potential mechanisms inducing spillover of an exploited planktivorous reef fish, *Naso vlamingii* (Acanthuridae), from a well-protected no-take reserve. Emigration of *N. vlamingii* from the reserve during the short term was examined from repeated visual censuses made over several days in the reserve and adjacent fished sites. The role of competition in spillover of *N. vlamingii* over the long term was investigated in a study of behavioural interactions among adults inside and outside the reserve. Additionally, experimental hook and line fishing for *N. vlamingii* was made outside the reserve at increasing distances from reserve boundaries to determine if gradual patterns of decreasing catch rate and mean size were present.

In Chapter 4, I attempted to quantify the contribution of spillover from a well-protected no-take reserve to the yield of a fishery at a small island (Apo Island, Philippines). Daily fishing effort and yield of local fishers in relation to the reserve were studied for eight months over two monsoonal seasons. Spatial and temporal patterns in fishing effort, yield (biomass and value), and catch rates were examined among major fishing grounds around the island and within a few hundred meters from the boundaries of the reserve.

The thesis concludes with a General Discussion (Chapter 5), synthesizing the results of the overall study, and suggesting future research in this area.



Chapter 2

Do gradients of abundance of fish exist across marine reserve boundaries?

Abstract. Gradients of abundance of fish across no-take marine reserve boundaries, from high inside the reserve to low outside, may indicate net export of adult fish, or spillover. Few studies have documented such gradients. Furthermore, these gradients may be caused by differences in fishing pressure or habitat inside and outside reserves. In this study, the spatial patterns of abundance of fish across two boundaries of each of two small reserves in the Philippines (protected for 20 and 15 years) were examined in 2002 to determine if gradients of abundance existed. Gradients of decreasing abundance from inside to outside the reserve were present for fish targeted by fisheries across one (but not the other) boundary of Apo Reserve (protected since 1982). Abundance of targeted fish declined sharply 50 m outside this boundary, but often declined more gradually up to 200 m outside the boundary, depending on the mobility of the fish group. The rates of decline in abundance of targeted fish across this Apo Reserve boundary were two to three times higher for sedentary than for vagile fish. This observation is consistent with predictions of how fish mobility would affect patterns of spillover. Changes in habitat did not affect the spatial pattern of abundance of targeted fish significantly across the Apo Reserve boundary. In contrast, fish not targeted by fisheries did not display a decrease in abundance across either boundary of Apo Reserve. Changes in habitat were more important in influencing their spatial patterns of abundance. However, decreasing gradients of abundance of targeted fish across the boundaries of Balicasag Reserve (protected since 1985) were absent. This result appears to be due to other factors, such as a general reduction in fishing levels outside the reserve since it was established. This study suggests that Apo Reserve has influenced the spatial pattern of abundance of targeted fish. The gradients across one boundary of this reserve were consistent with spillover. However, long-term studies of reserves that investigate changes in fish abundance, distribution and movement through time are required to determine if spillover could cause such patterns.

2.1 Introduction

A decreasing gradient of abundance of targeted fish across a reserve boundary, from high inside to low outside the reserve, may indicate net export of adult fish, or spillover (Rakitin & Kramer 1996). Net emigration of adult fish from the reserve, combined with fishing mortality outside the reserve, may produce such a decreasing pattern of abundance (Figure 2.1). For example, if fish density in the reserve has increased over time, some fish may relocate their home range to outside the reserve because density-dependent effects, such as competition for resources, may increase inside the reserve (Kramer & Chapman 1999, Sutherland et al. 2002). However, the number of relocated fish should decrease away from the reserve boundary (Kramer & Chapman 1999, Sutherland et al. 2002) (Figure 2.1a). Alternatively, a decreasing gradient of abundance may result from some fish that reside near the reserve boundary having home ranges that overlap the boundary (Kramer & Chapman 1999) (Figure 2.1b). Those fishes with home ranges located inside the reserve, but straddling the boundary, should decrease in abundance over time relative to fish completely inside the reserve, since they are sometimes exposed to fishing mortality. Fishes with home ranges located outside the reserve, but straddling the boundary, should increase in abundance over time relative to fish with home ranges completely outside the reserve, because they receive some protection during occasional movements into the reserve. However, habitat characteristics at the local scale (e.g. within a reef zone) may influence the distribution patterns of fish (Williams 1991). For example, habitat complexity (often associated with live hard coral cover in the tropics) and current strength may correlate positively with fish abundance, often because of greater availability of shelter sites in the former instance, and enhanced food availability (e.g. plankton and planktivorous prey) in the latter (Luckhurst & Luckhurst 1978 Carpenter et al. 1981, Thresher 1983, Roberts & Ormond 1987, Hixon & Beets 1989, Connell & Kingsford 1998). Thus, gradients of decreasing abundance across reserve boundaries may reflect better habitat inside than outside reserves, rather than effects of protection from fishing in the reserve.

Few studies have examined spatial patterns of abundance of fish near boundaries of reserves. Russ and Alcala (1996a) showed that the density of large predatory reef fish became significantly higher close to (within 200 m) than further from the southern

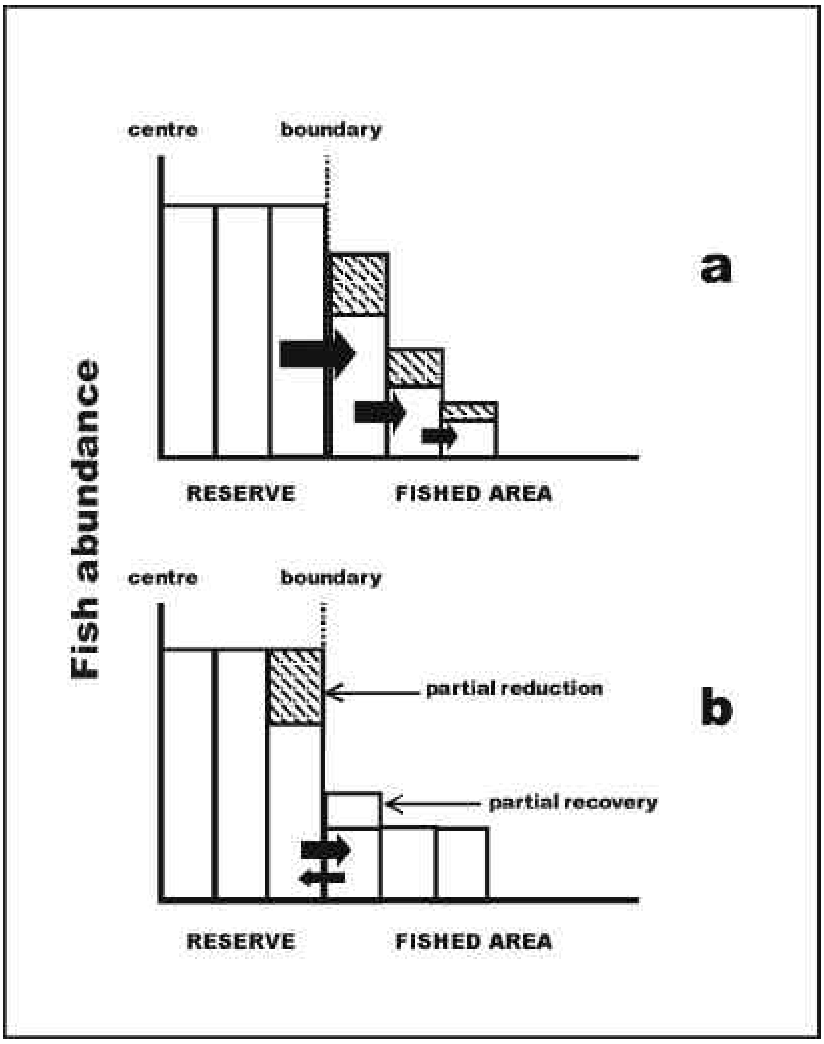


Figure 2.1. Gradients of decreasing fish abundance across boundaries of no-take reserves may result from net movement of fish from a reserve due to increased competition for resources (black arrows, a). Alternatively, gradients may be produced by short-term movements of fish (black arrows, b) with home range straddling the reserve boundary. In the case of straddling home ranges, fish with home ranges centred inside the reserve may sometimes be exposed to fishing mortality, while those centred outside the reserve will occasionally be protected. The mechanism for model a is density-dependent emigration (a long-term process). The mechanism for model b is movement by fishes within their home ranges (a short-term process). Shading represents reduction in abundance due to fishing.

boundary of Apo Reserve in the Philippines during a decade of protection. Russ et al. (2003, 2004) reported a similar pattern for the distribution of surgeonfishes and jacks near the same boundary – a pattern that developed over a period of almost 20 years. They also reported higher catch rates of surgeonfishes closer to the southern and northern boundaries of Apo Reserve than further away (Russ et al., 2003, 2004). However, these studies did not account for differences in habitat between the reserve and the fished sites. These patterns could have been due to spillover, or to habitat differences, if habitat gradually improved over time just outside the reserve. McClanahan and Mangi (2000) found that catch rates and fish density decreased with distance from the boundaries of the Mombasa Marine Park in Kenya up to 5 km from the boundary. They did not find the same relationship between benthic habitat parameters and distance from the reserve, and reasoned that the pattern in fish abundance was probably due to spillover rather than habitat effects.

Rakitin and Kramer (1996) found a negative relationship between catch rate of traps and distance from the centre of the Barbados Marine Reserve, up to 2 km outside the boundary. However, the density of trappable fish, estimated from visual census, did not decrease with distance from the reserve centre. Also at the Barbados reserve, Chapman and Kramer (1999) showed that the density of trappable fish decreased with distance from the reserve centre. However, they found that this relationship was not significant after statistically controlling for habitat effects. In a tagging study at the same location, they did not detect movement of fish from the reserve to the fished area (Chapman & Kramer 2000). Studies at the Barbados reserve suggest that habitat differences, rather than spillover, mainly account for the observed spatial patterns of fish abundance.

Given the small number of studies on spatial patterns of abundance of fish near reserve boundaries, it is still unclear whether a decreasing gradient of fish abundance (or indices of abundance such as catch rate) across a reserve boundary indicates spillover. One prediction of the few empirical and theoretical studies of spillover is that decreasing gradients of abundance would likely be limited to tens of meters to a few kilometres from the reserve boundary, depending on the species (Russ 2002). A further prediction is that the slope of gradients of abundance across reserve boundaries would vary depending on the catchability and mobility of fish. Assuming equal mobility, the gradient would be steeper for fish more than less vulnerable to fishing gears (Rakitin & Kramer 1996). No gradient should be observed if fish are not vulnerable to the fishery

(Rakitin & Kramer 1996). Assuming equal catchability, the gradient would be steeper for more sedentary than less sedentary species (Rakitin & Kramer 1996). Highly vagile species may display a very weak gradient of abundance across reserve boundaries (Rakitin & Kramer 1996). However, few studies have empirically demonstrated patterns of declining abundance of fish across reserve boundaries, let alone related rates of decline to aspects of mobility or vulnerability of the target species.

This study investigated the spatial pattern of abundance of reef fish across two boundaries of each of two small reserves in the Philippines that have been protected for 20 and 15 years. The aim was to quantify patterns of abundance of fish targeted by fisheries across reserve boundaries, and to relate these patterns to habitat characteristics across the boundaries. It was predicted that if gradients of abundance existed, the slopes of the gradients would differ among sedentary, vagile and highly vagile targeted fish. No decrease in the abundance of fish not targeted by fisheries was expected across reserve boundaries.

2.2 Materials and Methods

The study was conducted at two small islands in the central Philippines (Figure 2.2). The first was Apo (9? 4’ N, 123? 17’ E), a high volcanic island with an area of about 0.70 km^2. Apo is surrounded by a narrow (100 m wide or less) fringing reef with an area of about 0.54 km^2 to the 20 m isobath. The eastern and southeastern portions of the reef surrounding Apo have the most extensive live coral cover. A 450 m long no-take reserve on the southeastern side of Apo has been protected effectively since 1982 (Russ & Alcala 1999) (Figure 2.2). The second was Balicasag (9? 31’ N, 123? 41’ E), a low coralline island with an area of about 0.20 km^2. Balicasag has a fringing reef area of about 0.3 km^2 to the 25 m isobath. The western and southern sides of the reef surrounding Balicasag have the most extensive live coral cover. A 650 m long no-take reserve on the southwestern side of Balicasag has been protected effectively since 1985 (Figure 2.2). Apo and Balicasag have communities of 700 to 800 people that engage in artisanal and subsistence fishing using methods non-destructive to the coral habitat. The no-fishing rule is strictly enforced in the reserves of the two islands, with good compliance from local people (Wells & White 1995, Russ & Alcala 1999, White et al. 2002). However, collection of deep-water ornamental shells is allowed in Balicasag Reserve (R. Abesamis, *personal observation*).

Fifteen 50 m x 10 m transects were surveyed on the shallow reef slope across each of the two reserves, and extending up to 200-400 m outside the lateral boundaries of the reserve (Figure 2.2). Similarly, fifteen transects were surveyed across a control (a fished area with dimensions equal to the reserve) at each island, and extending up to 200-400 m outside the lateral boundaries of the control (Figure 2.2). The transects were positioned sequentially at a depth of 10-12 m, parallel to the shore. They were placed at least 10 m apart at Apo and at least 20 m apart at Balicasag. The transects that crossed the reserve at each of the two islands were generally located in areas with high coral cover, high structural complexity and steep slope (40-90° decline). In contrast, the transects that crossed the controls at each of the two islands were generally located in areas with low coral cover and shallow slope (20-40° decline), dominated by sand or seagrass.

Fish and habitat surveys were made along these transects by two divers using SCUBA. Surveys were made in November and December 2002, during the northeast

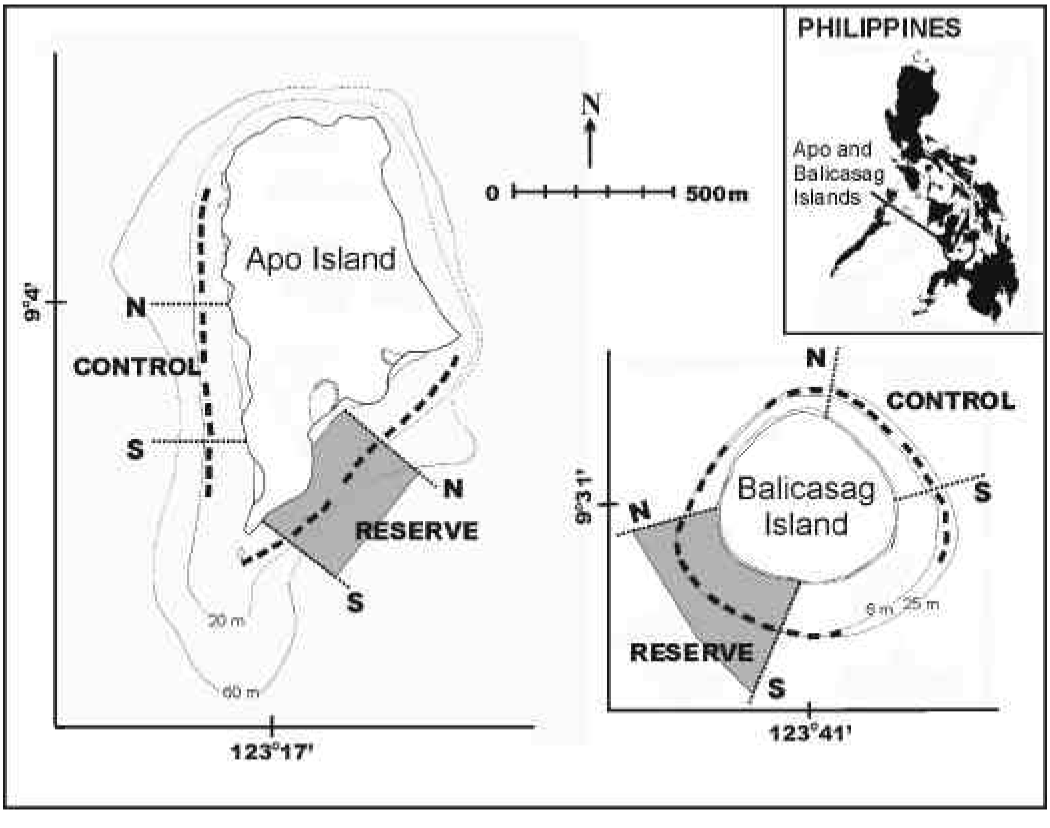


Figure 2.2. Maps of Apo and Balicasag islands showing the no-take reserve (shaded) and control area (open to fishing) at each island. Black rectangles show the approximate positions of individual 50 x 10 m transects surveyed visually in the study. N – northern boundaries, S – southern boundaries

monsoon. Fish size estimation by one diver (R.A.A.) was calibrated by estimating lengths of PVC pipes of 35 different sizes underwater (size range: 10-78 cm). Surveys commenced after size estimation error was consistently within ? 2 cm. Three transects were completed per dive. The first diver (R.A.A.) recorded the numbers and sizes [total length (TL)] of targeted fish (95 spp. from 13 families, Table 2.1) that were seen within 5 m either side of each transect, excluding targeted fish less than 10 cm TL. The second diver followed the first, laying the 50 m transect line. The first diver also scored current strength according to a 3-point scale (1 – weak, 2 – moderate, 3 – strong) every 10 m along each transect.

After surveying reef fish along three transects, the divers swam back along the three 50 m tapes left on the bottom. The second diver then classified the benthic habitat every 2 meters (from 48 to 0 m) along each transect line as either live hard coral (massive or complex lifeforms), dead coral, seagrass, sand, rubble, rock or ‘others’. Simultaneously, the first diver recorded the abundance of non-targeted fish (4 species of Pomacentridae, 16 species of Chaetodontidae, Table 2.1) that were seen within 2.5 m either side of the transect, while reeling-in the transect tape. The first diver also scored the rugosity of the reef substratum [0 (least rugose) to 4 (most rugose)] and the steepness of the reef slope [0 (horizontal) to 4 (vertical)] visually for every 10 m long x 5 m wide area along the transect (5 estimates per transect). Dives averaged 70 minutes duration. Minimum visibility on all dives was 25 m.

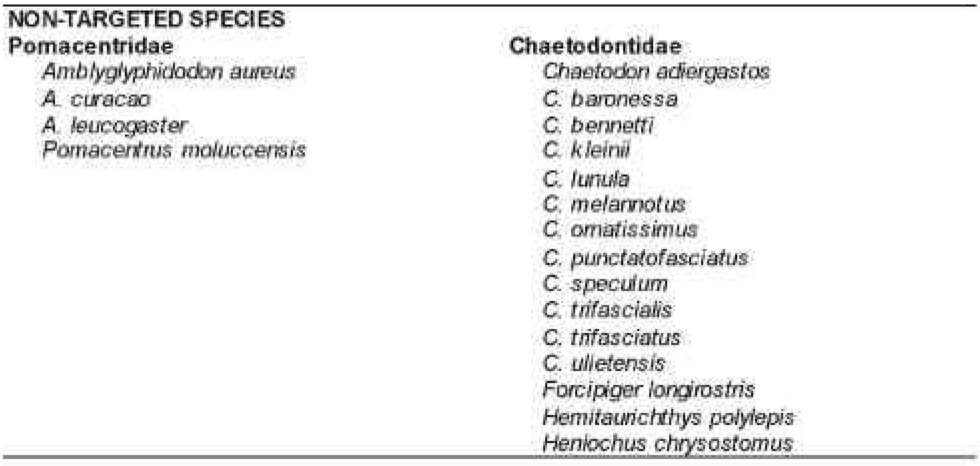
All transects were surveyed twice. The order of survey dives amongst transects in an island was random (weather permitting). However, all transects at an island were surveyed once before resampling at that island commenced (Apo – 1st survey: 21-25 November, 2nd survey: 25-29 November, 2002; Balicasag – 1st survey: 2-6 December, 2nd survey: 6-9 December, 2002). It was possible to place transects within 5-10 m of the first sampling position by making use of permanent mooring buoys, natural underwater features, and a GPS receiver.

Density of targeted fish was expressed as number per 500 m^2, density of non-targeted fish as number per 250 m^2. Biomass of targeted fish per transect was estimated as the sum of individual weights of fish, in kg, per 500 m^2. Individual weights of fish

Table 2.1. Targeted and non-targeted fish species recorded in the underwater visual censuses. Mobility groupings for targeted fish: S – sedentary; V – vagile; HV – highly vagile.

TARGETED SPECIES	
Acanthuridae	
Acanthurus blochii	V
A. dussumieri	V
A. fowleri	V
A. lineatus	V
A. mata	HV
A. nigricans	V
A. nigricauda	V
A. olivaceus	V
A. pyroferus	V
A. thompsoni	HV
Ctenochaetus. binotatus	V
C. striatus	V
C. strigosus	V
C. tominiensis	V
Naso brachycentron	HV
N. brevirostris	HV
N. hexacanthus	HV
N. lituratus	V
N. lopezi	HV
N. minor	HV
N. thynnoides	HV
N. unicornis	HV
N. vlamingii	HV
Carangidae	
Carangoides ferdau	HV
C. gymnostethus	HV
Caranx ignobilis	HV
C. melampygus	HV
C. sexfasciatus	HV
Caranx sp.	HV
Elegatis bipinnulatus	HV
Scomberoides lysan	HV
Ephippidae	
Platax teira	V
Haemulidae	
Plectorhincus chaetodonoides	S
P. goldmani	S
Kyphosidae	
Kyphosus cinerascens	HV
K. vaigensis	HV
Labridae	
Cheilinus undulatus	V
Hemigymnus fasciatus	V
H. melapterus	V
Lethrinidae	
Lethrinus erythracanthus	V
L. erythropterus	V
L. harak	V
L. obsoletus	V
Monotaxis grandoculis	V
Lutjanidae	
Aphareus furca	HV
Lutjanus argentimaculatus	V
L. biguttatus	V
Lutjanus bohar	V
L. decussatus	V
L. ehrenbergi	V
L. fulviflamma	V
L. gibbus	V
L. monostigma	V
L. rivulatus	V
Macolor macularis	V
M. niger	V
Mullidae	
Mulloidichthys flavolineatus	V
Parupeneus barberinus	V
P. bifasciatus	V
P. multifasciatus	V
Scaridae	
Calatomus carolinus	V
Cetoscarus bicolor	V
Chlorurus bleekeri	V
C. bowersi	V
C. microrhinos	V
Scarus altipinnis	V
S. dimidiatus	V
S. festivus	V
S. flavipectoralis	V
S. forsteni	V
S. frenatus	V
S. ghobban	V
S. niger	V
S. oviceps	V
S. prasiognathus	V
S. psittacus	V
S. schlegeli	V
S. sordidus	V
S. spinus	V
S. tricolor	V
Scarus sp.	V
Scombridae	
Gymnosarda unicolor	HV
Serranidae	
Anyperodon leucogrammicus	S
Cephalopholis argus	S
C. boenak	S
C. cyanostigma	S
C. sexmaculatus	S
Epinephelus merra	S
Plectropomus oligacanthus	V
Variola albimarginata	V
Siganidae	
Siganus argenteus	V
S. fuscescens	V
S. guttatus	V
S. punctatissimus	V
S. virgatus	V
S. vulpinus	V

Table 2.1. Continued



NON-TARGETED SPECIES	
Pomacentridae	**Chaetodontidae**
Amblyglyphidodon aureus	*Chaetodon adiergastos*
A. curacao	*C. baronessa*
A. leucogaster	*C. bennetti*
Pomacentrus moluccensis	*C. kleinii*
	C. lunula
	C. melannotus
	C. ornatissimus
	C. punctatofasciatus
	C. speculum
	C. trifascialis
	C. trifasciatus
	C. ulietensis
	Forcipiger longirostris
	Hemitaurichthys polylepis
	Heniochus chrysostomus

were estimated from published length-weight relationships (Froese & Pauly 2003). When length-weight relationships were not available for a species, those of a closely related species were used. Targeted fish were classified into three mobility guilds (Table 2.1). Mobility guilds were broad due to the paucity of information on movement of reef fishes. The 'sedentary' guild included species of Haemulidae and Serranidae. The 'vagile' guild included Acanthuridae (*Acanthurus* and *Ctenochaetus* spp.), Ephippidae, Labridae, Lethrinidae, Lutjanidae, Mullidae, Scaridae and Siganidae. The 'highly vagile' guild included Acanthuridae (*Naso* spp.), Kyphosidae, Carangidae and Scombridae. Mobility groupings were based mainly on personal observations and the few published studies of the home range of certain species (Holland et al. 1993, 1996, Zeller & Russ 1998, Meyer et al. 2000, Eristhee & Oxenford 2001). Available information on movement were often generalised for closely related species or families. Very large schools of targeted fish (e.g. Carangidae) that could not be counted and have their sizes estimated accurately were excluded from the data used here.

The percent cover of benthic categories was calculated by dividing the number of points for each habitat component by 25 (the total number of points surveyed per line-intercept transect). The mean reef rugosity, mean reef steepness and mean current strength, were calculated as the average of the five estimates per transect. Most benthic habitat variates were correlated with each other. Proportion of live hard coral cover (both massive and complex lifeforms), mean reef rugosity, and mean reef steepness were positively correlated. However, proportion of hard coral cover, rugosity and reef steepness were negatively correlated with percent cover of sand, rubble, rock and seagrass. Thus, description of the benthic habitat for each transect was estimated as a single habitat complexity index (HCI):

HCI = (proportion of live hard coral cover + 1)*(mean rugosity + 1)*(mean reef steepness + 1).

This index ranges from 1-50. Higher values indicate steeper reef faces with high hard coral cover and rugosity. Lower values indicate relatively flat expanses of sand, rubble, rock or seagrass, with low hard coral cover.

Repeated measures ANOVA was used to determine whether density or biomass of fish differed significantly between inside and outside reserves, since the same transects were surveyed twice. Fish density and biomass inside the reserve (from 6 transects at Apo and 8 transects at Balicasag, all sampled twice) were compared with those of all replicates in fished areas outside of the reserve (24 transects at Apo, 22 transects at Balicasag, all sampled twice) at each island.

Linear regression was used to determine if a significant relationship existed between mean fish density, or biomass, and distance from the centre of reserves and controls. These regressions were performed separately for the two lateral boundaries of each reserve and for the lateral boundaries of each control. Where significant trends of decreasing density or biomass of targeted fish from different mobility guilds were found across a reserve boundary, comparisons of regression slopes were made using the procedure described by Zar (1999). In order to compare the regression slopes of two mobility guilds, mean density and biomass on transects were expressed as percent change from the mean density or biomass of the first transect nearest to the centre of the reserve. HCI, dead coral cover, and current strength at each transect were also regressed against distance from the centre of reserves and controls to assess if significant trends in habitat variates across lateral boundaries correlated with trends in fish abundance.

Stepwise multiple regression was used to assess if reserve protection or habitat characteristics were a more important influence on the spatial patterns of fish abundance across a reserve. Fish density and biomass were regressed against protection status (reserve vs. fished area), HCI, percent dead coral cover, and current strength. Both the first and second measurements of fish and habitat variates on each transect were used in multiple regression across each reserve and control (i.e., n = 30, or 2 measurements x 15 transects, for each regression across a reserve or control). For protection status, a dummy variable of ‘2’ was assigned to transects inside reserves, and ‘1’ for those in fished areas. Hence, a positive relationship between fish abundance and protection status would indicate a positive effect of the reserve. However, to see how well the analysis could distinguish between a real reserve and a fished control, a dummy variable of ‘2’ was also assigned to transects inside the fished controls (see Figure 2.2).

2.3 Results

At Apo, but not at Balicasag, targeted fish had a significantly higher density (island: n.s.; protection: $F_{1,56}$ = 20.52, p ? 0.001; island x protection: $F_{1,56}$ = 18.40, p ? 0.001) and biomass (island: n.s.; protection: $F_{1,56}$ = 18.94, p ? 0.001; island x protection: $F_{1,56}$ = 18.13, p ? 0.001) inside the reserve than in fished areas. Sampling time did not have a significant effect on density or biomass. The reserve at Apo had 2.5 times the density [reserve: 67.9 ? 8.8 (S.E.), fished areas: 26.9 ? 1.9 fish 500 m^{-2}] and 3.6 times the biomass (reserve: 89.1 ? 17.9, fished areas: 24.8 ? 3.6 kg 500 m^{-2}) of targeted fish in fished areas. No significant gradients of abundance (density and biomass) of targeted fish were found across the boundaries of reserves and controls at both islands, except for the northern boundary of the reserve at Apo. A significant decrease of density of targeted fish (Figure 2.3, mean density = -0.11 x distance from reserve centre + 47.17, r^2 = 0.63, $F_{1,7}$ = 12.07, p = 0.01) and biomass (mean biomass = -0.22 x distance from reserve centre + 65.15, r^2 = 0.67, $F_{1,7}$ = 14.36, p = 0.007) was present across this boundary. However, across the same reserve boundary, there were no significant spatial trends of habitat complexity, dead coral cover or current strength (Table 2.2).

Across Apo Reserve, density and biomass of targeted fish were correlated positively with protection status (Table 2.3). Habitat variates had minimal effects on the spatial pattern of abundance of targeted fish (Table 2.3). However, across Balicasag Reserve and the control transects at both islands, density and biomass of targeted fish were not correlated significantly with protection status. Habitat variates were a more important influence on biomass of targeted fish, but not density, across these locations (Table 2.3).

No decreasing gradients of abundance were detected for density of non-targeted fish across any reserve or control boundary (Figure 2.4). This occurred even though density was higher inside reserves than in fished areas, by a factor of 1.2 at Apo and 1.5 at Balicasag (island: $F_{1,56}$ = 14.94, p ? 0.001; protection: $F_{1,56}$ = 4.10, p ? 0.05; island x

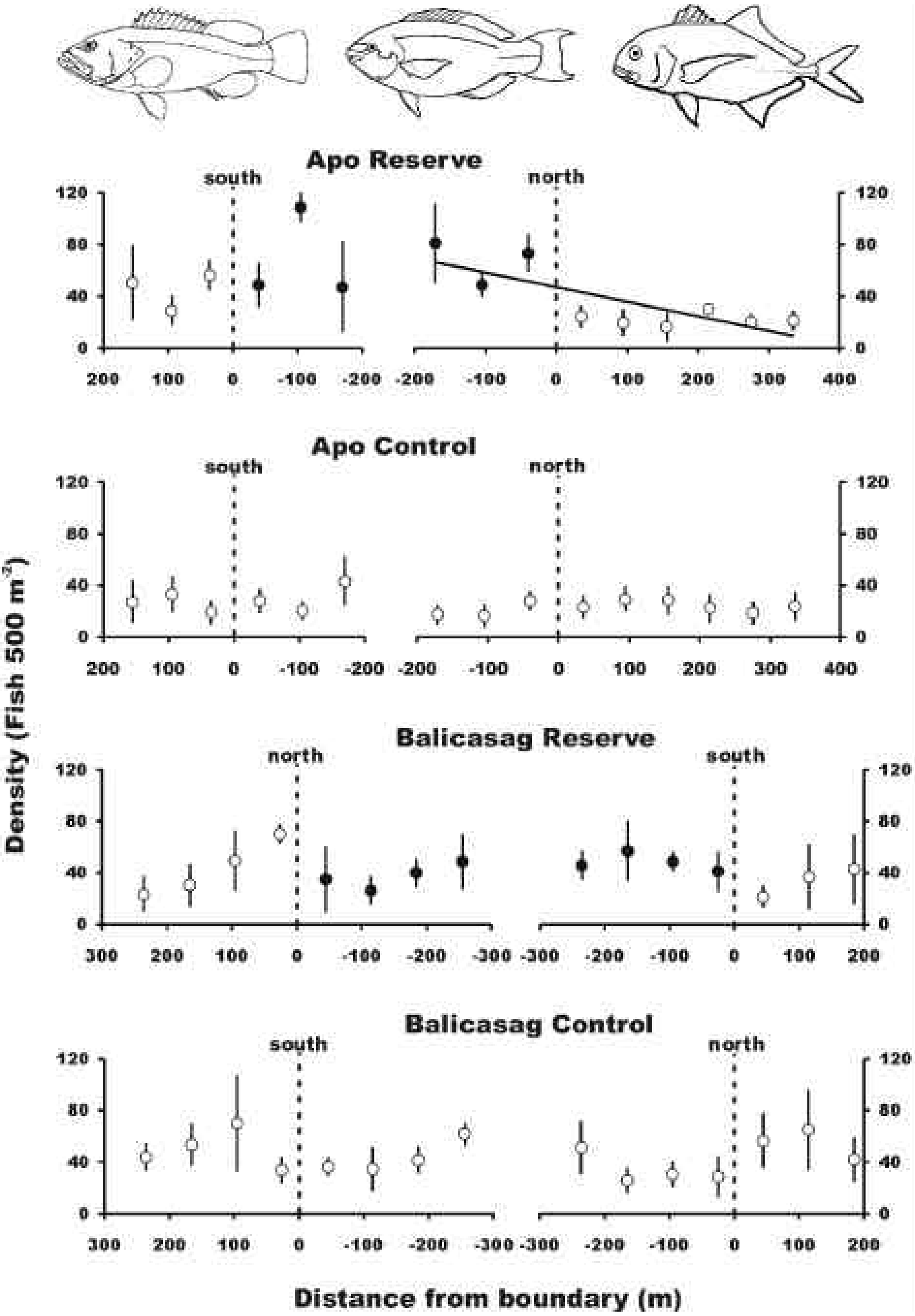


Figure 2.3. Mean density of targeted fish across the northern and southern boundaries (broken vertical lines) of no-take reserves and fished controls at Apo and Balicasag islands (see Figure 2.2). Filled and open circles indicate mean density in reserve and fished areas, respectively. A regression line is shown where there was a significant relationship between mean density and distance from the centre of the reserve or control. Error bars are ? 1 SE.

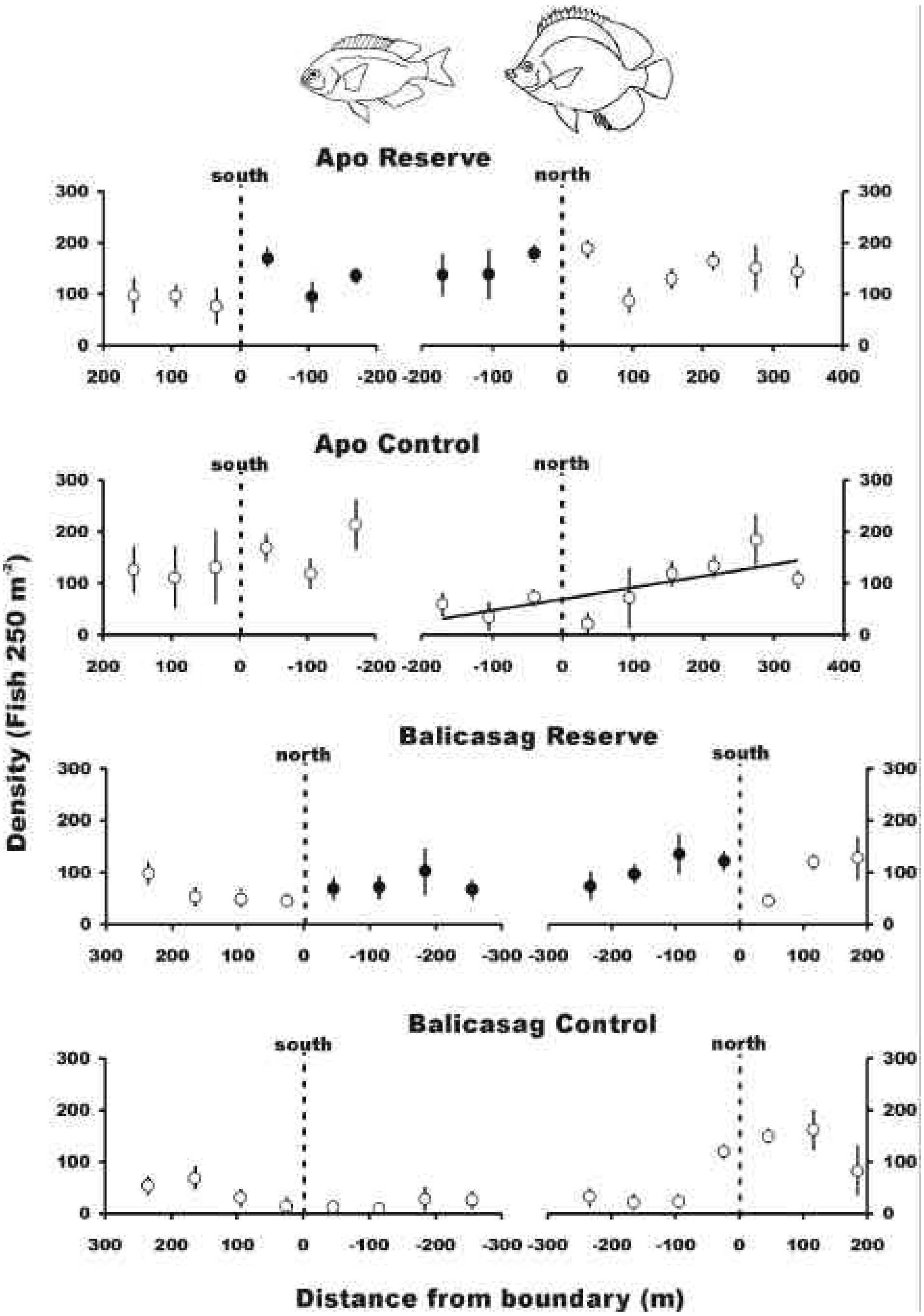


Figure 2.4. Mean density of non-targeted fish across the northern and southern boundaries (broken vertical lines) of no-take reserves and fished controls at Apo and Balicasag islands (see Figure 2.2). Filled and open circles indicate mean density in reserve and fished areas, respectively. A regression line is shown where there was a significant relationship between mean density and distance from the centre of the reserve or control. Error bars are ? 1 SE.

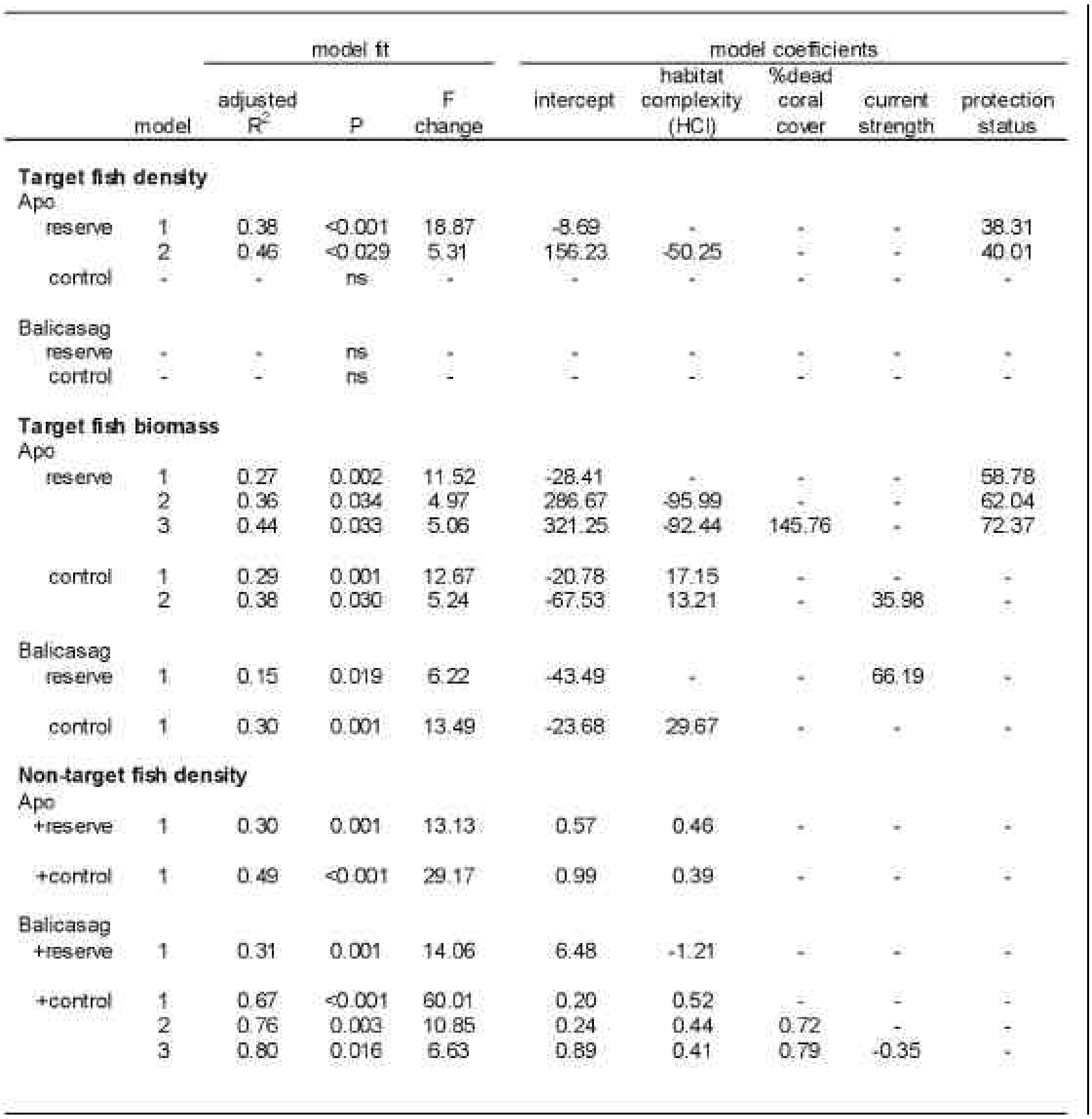
Table 2.2. Regression analyses of habitat variates across each boundary (north and south) of reserves and controls at Apo and Balicasag islands (see Figure 2.2). Habitat variates were regressed against distance from centre of reserves or controls. *p ?0.05, **p ?0.01, ***p ?0.001, ns – not significant at ? = 0.05. (+) and (-) indicates positive and negative relationship of habitat variate with distance from centre of reserve or control. Habitat complexity, % dead coral cover, and current strength were ln (x+1), $\sin^{-1}(x)^{1/2}$ and $(x)^{1/2}$ transformed, respectively.

	Apo Reserve		Apo Control		Balicasag Reserve		Balicasag Control	
	south (1,4 df)	north (1,7 df)	south (1,4 df)	north (1,7 df)	north (1,6 df)	south (1,5 df)	south (1,6 df)	north (1,5 df)
Habitat complexity (HCI)								
R^2	0.61	0.06	0.73	0.52	0.29	0.41	0.76	0.92
F	6.34 ns	0.43 ns	11.03* (-)	7.65* (+)	2.47 ns	3.56 ns	19.07** (+)	61.20*** (+)
% dead coral								
R^2	0.02	0.03	0.55	0.14	0.27	0.22	0.19	0.001
F	0.79 ns	0.68 ns	4.93 ns	1.17 ns	2.22 ns	1.43 ns	0.29 ns	0.006 ns
Current strength								
R^2	0.74	0.02	0.17	0.35	0.22	0.62	0.05	0.26
F	11.52*(-)	0.14 ns	0.80 ns	3.68 ns	1.65 ns	8.31*(+)	0.30 ns	1.71 ns

protection: n.s.). The only significant spatial pattern detected for non-targeted fish was a gradient of increasing density across the northern boundary of the control at Apo (Figure 2.4, mean density = -0.22 x distance from centre of control + 69.42, $r^2 = 0.57$, $F_{1,7} = 9.12$, $p = 0.019$). This result may be influenced by increasing habitat complexity across this control boundary (Table 2.2). Across all reserves and controls, patterns of density of non-targeted fish were accounted for mainly by habitat variates, and not by protection status (Table 2.3).

Decreasing trends in the density and biomass of sedentary, vagile and highly vagile targeted fish were present across the northern boundary of Apo Reserve only (Figure 2.5). The pattern of decline in density and biomass across this boundary was steeper for sedentary than for vagile fish (Figure 2.6, regression slopes for density – sedentary fish: -0.32, vagile fish: -0.09; regression slopes for biomass – sedentary fish: -0.27, vagile fish: -0.15). However, the regression slopes for sedentary and vagile fish were significantly different only for density (t-test, $t_{14\ df} = 2.15$, $P < 0.05$). Density and biomass of sedentary and vagile fish across the transects at Apo Reserve were not correlated with habitat variates (Table 2.4). However, density of sedentary fish, and biomass of sedentary and vagile fish were correlated positively with protection status (Table 2.4). Highly vagile fish, however, had a relatively steep gradient (Figure 2.5).

Table 2.3. Results of stepwise multiple regressions of density and biomass of fish versus habitat variates and protection status across reserve and control transects (see Figure 2.2). Significant models of the stepwise regression at a reserve or control are numbered consecutively. Habitat complexity, % dead coral cover, and current strength were ln (x+1), $\sin^{-1}(x)^{½}$, and $(x)^{½}$ transformed, respectively. + indicates $\log_{10}$ (x+1) transformation of dependent variate.



		model fit			model coeficients				
	model	adjusted R^2	P	F change	intercept	habitat complexity (HCI)	%dead coral cover	current strength	protection status
Target fish density									
Apo									
reserve	1	0.38	<0.001	18.87	-8.69	-	-	-	38.31
	2	0.46	<0.029	5.31	156.23	-50.25	-	-	40.01
control	-	-	ns	-	-	-	-	-	-
Balicasag									
reserve	-	-	ns	-	-	-	-	-	-
control	-	-	ns	-	-	-	-	-	-
Target fish biomass									
Apo									
reserve	1	0.27	0.002	11.52	-28.41	-	-	-	58.78
	2	0.36	0.034	4.97	286.67	-95.99	-	-	62.04
	3	0.44	0.033	5.06	321.25	-92.44	145.76	-	72.37
control	1	0.29	0.001	12.67	-20.78	17.15	-	-	-
	2	0.38	0.030	5.24	-67.53	13.21	-	35.98	-
Balicasag									
reserve	1	0.15	0.019	6.22	-43.49	-	-	66.19	-
control	1	0.30	0.001	13.49	-23.68	29.67	-	-	-
Non-target fish density									
Apo									
+reserve	1	0.30	0.001	13.13	0.57	0.46	-	-	-
+control	1	0.49	<0.001	29.17	0.99	0.39	-	-	-
Balicasag									
+reserve	1	0.31	0.001	14.06	6.48	-1.21	-	-	-
+control	1	0.67	<0.001	60.01	0.20	0.52	-	-	-
	2	0.76	0.003	10.85	0.24	0.44	0.72	-	-
	3	0.80	0.016	6.63	0.89	0.41	0.79	-0.35	-

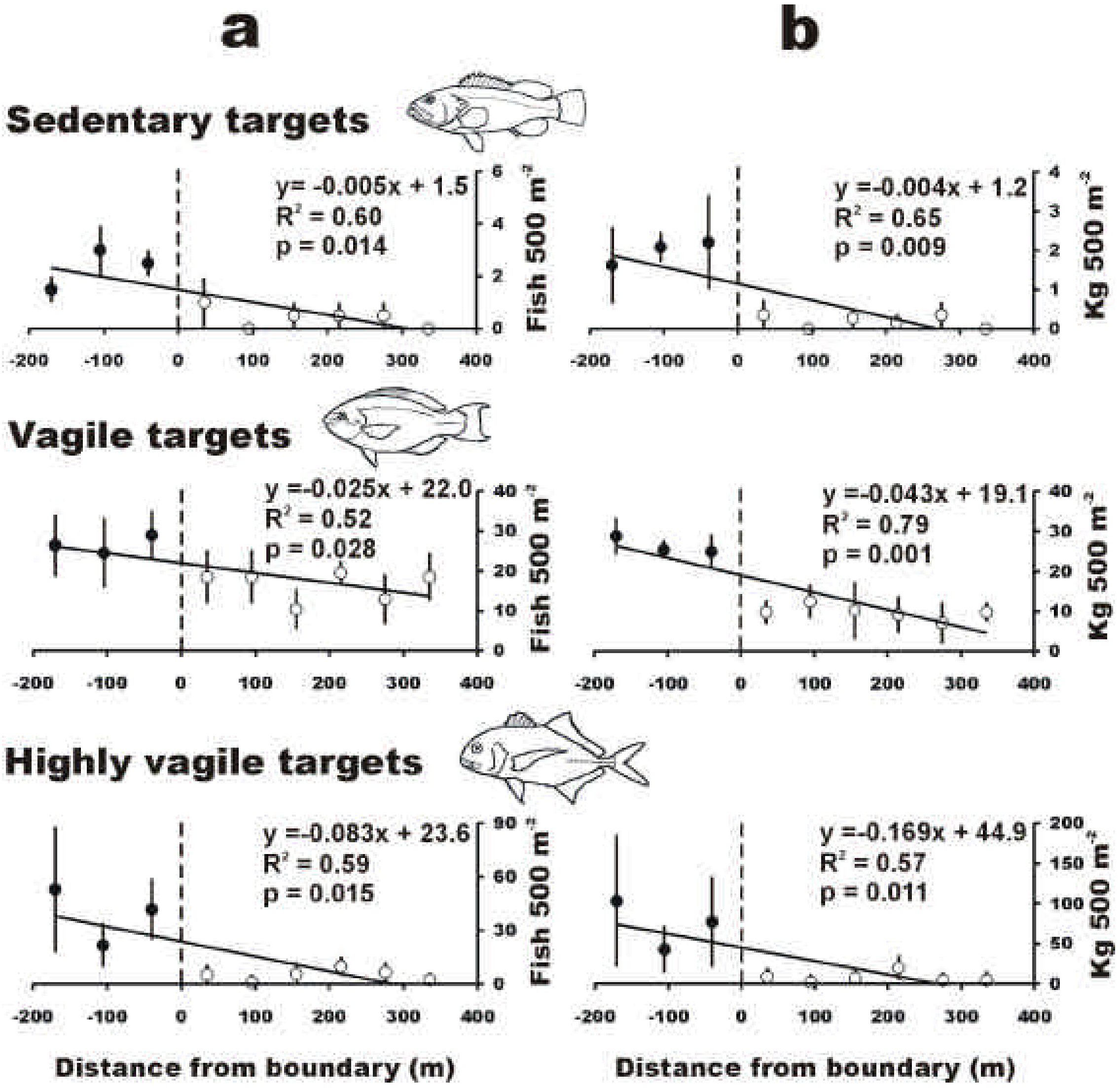


Figure 2.5. The spatial distribution of mean density (a) and mean biomass (b) of three different mobility categories of targeted fish across the northern boundary (broken vertical lines) of Apo Reserve (see Figure 2.2). Filled and open circles indicate mean density or biomass in reserve and fished areas, respectively. Error bars are ? 1 SE.

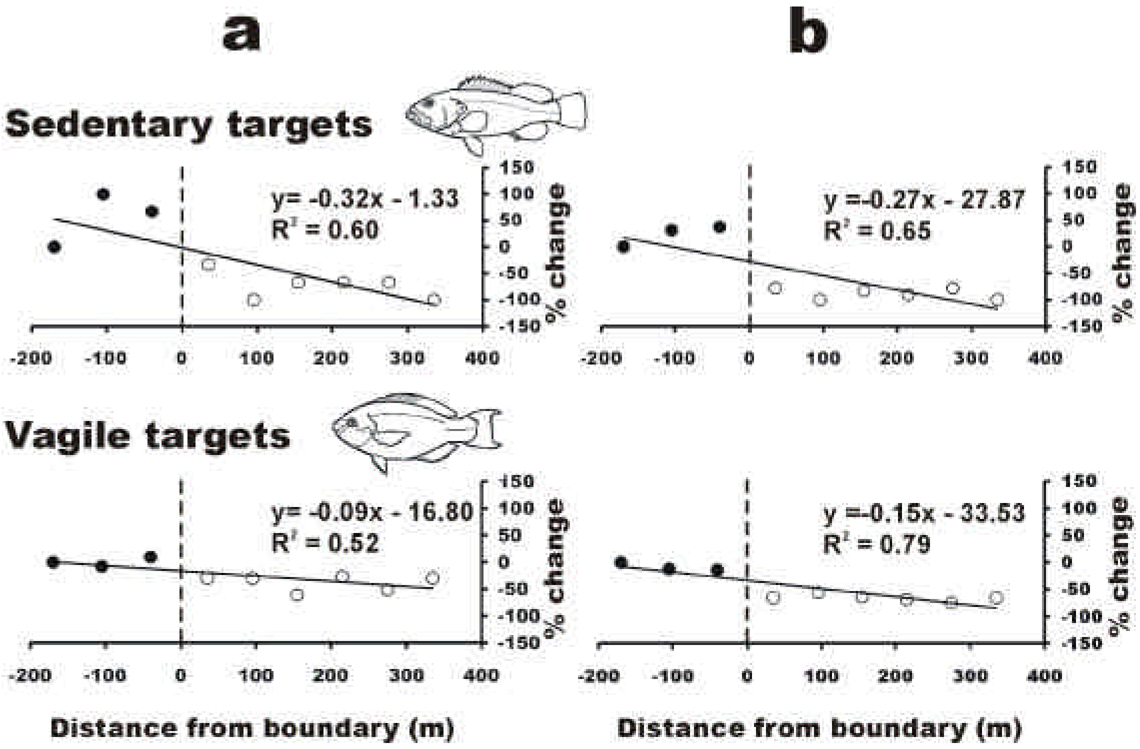
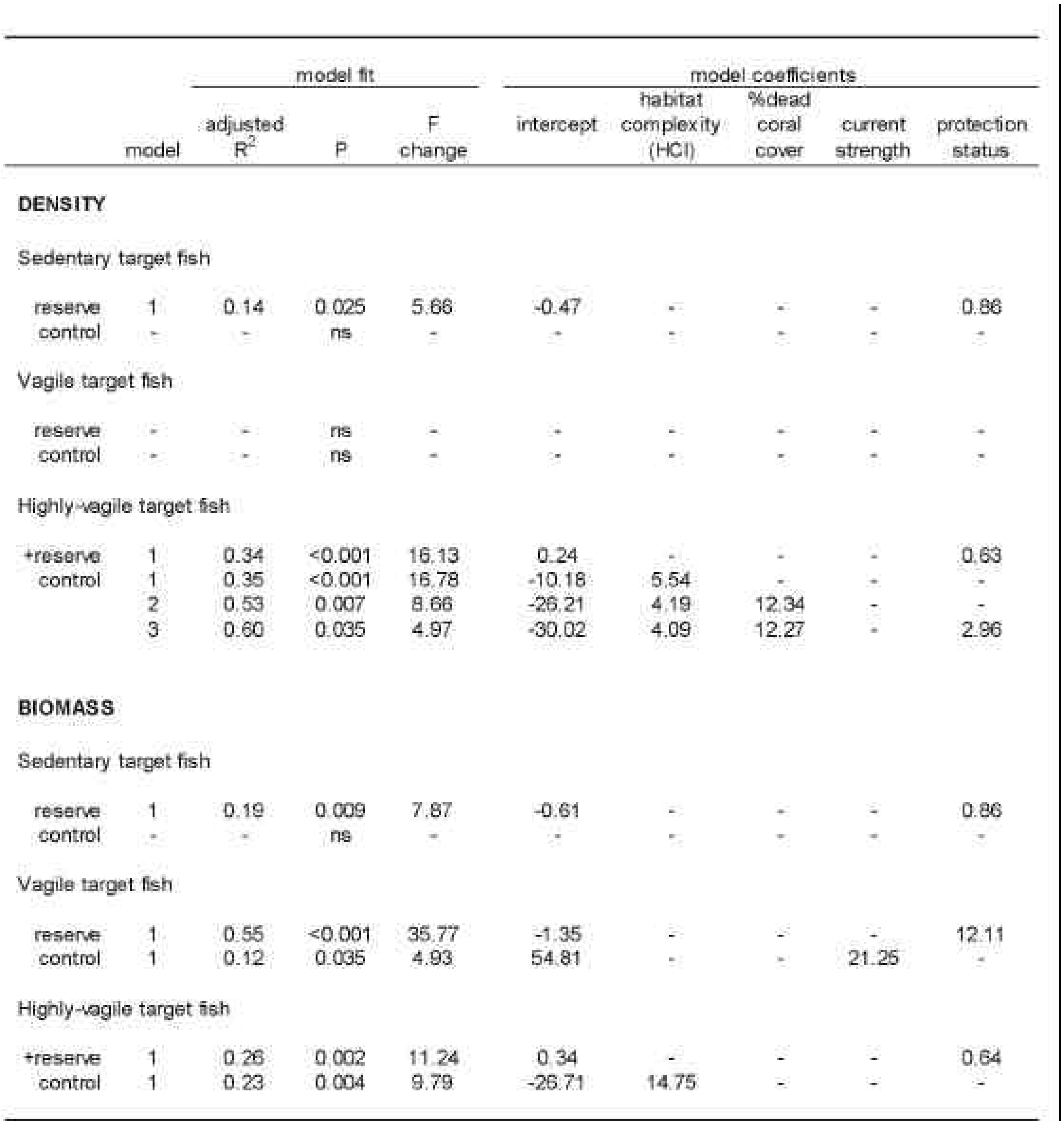


Figure 2.6. The spatial distribution of mean density (a) and mean biomass (b) of sedentary and vagile targeted fish across the northern boundary (broken vertical lines) of Apo Reserve (see Figure 2.2). Density and biomass at each transect are expressed as percent change in mean density or biomass from the transect nearest the reserve centre (-170 m).

Table 2.4. Results of stepwise multiple regressions of density and biomass of different mobility guilds of target fish versus habitat variates and protection status across reserve and control transects at Apo Island. Significant models of the stepwise regression at the reserve and control are numbered consecutively. Habitat complexity, dead coral cover, and current strength were ln (x+1), $\sin^{-1}(x)^{½}$, and $(x)^{½}$ transformed, respectively. + indicates $\log_{10}$ (x+1) transformation of dependent variate.



		model fit			model coefficients				
	model	adjusted R^2	P	F change	intercept	habitat complexity (HCI)	%dead coral cover	current strength	protection status
DENSITY									
Sedentary target fish									
reserve	1	0.14	0.025	5.66	-0.47	-	-	-	0.86
control	-	-	ns	-	-	-	-	-	-
Vagile target fish									
reserve	-	-	ns	-	-	-	-	-	-
control	-	-	ns	-	-	-	-	-	-
Highly-vagile target fish									
+reserve	1	0.34	<0.001	16.13	0.24	-	-	-	0.63
control	1	0.35	<0.001	16.78	-10.18	5.54	-	-	-
	2	0.53	0.007	8.66	-26.21	4.19	12.34	-	-
	3	0.60	0.035	4.97	-30.02	4.09	12.27	-	2.96
BIOMASS									
Sedentary target fish									
reserve	1	0.19	0.009	7.87	-0.61	-	-	-	0.86
control	-	-	ns	-	-	-	-	-	-
Vagile target fish									
reserve	1	0.55	<0.001	35.77	-1.35	-	-	-	12.11
control	1	0.12	0.035	4.93	54.81	-	-	21.25	-
Highly-vagile target fish									
+reserve	1	0.26	0.002	11.24	0.34	-	-	-	0.64
control	1	0.23	0.004	9.79	-26.71	14.75	-	-	-

Their density and biomass across transects at Apo Reserve was correlated positively with protection status (Table 2.4).

2.4 Discussion

The aim of this study was to determine if gradients of decreasing abundance of targeted fish occurred across the boundaries of two Philippine marine reserves. Such patterns may indicate spillover. However, decreasing gradients of abundance may also occur even if net emigration from the reserve is absent. The latter type of gradient could be due to better fish habitat in reserve than fished areas.

If spillover causes gradients of decreasing abundance of targeted fish from reserve to fished areas, then such gradients are more likely to occur in reserves that have been protected effectively for a long time, or reserves where the abundance of targeted fish has increased substantially, due to protection (Russ 2002). However, of the two reserves that were studied here, greater density and biomass of targeted fish in the reserve relative to the fished areas was found at only one – Apo Reserve. Furthermore, linear regression detected gradients of decreasing abundance of targeted fish across only the northern boundary of this reserve. It is unlikely, however, that habitat explains the spatial pattern of abundance of targeted fish across the northern boundary of Apo Reserve. Habitat variates did not show any significant patterns of change across the boundary. Furthermore, the results suggested that protection, rather than habitat, was more important in affecting the patterns of spatial abundance of targeted fish at Apo Reserve. In contrast, non-targeted fish did not have a pattern of decreasing abundance across this boundary. Habitat variates, rather than reserve protection, influenced the spatial patterns of abundance of non-targeted fish more strongly across Apo Reserve.

It could be suggested that the spatial pattern of abundance of targeted fish across the northern boundary of Apo Reserve does not display a decreasing gradient (Figure 2.3). Density (and biomass) seemed to decrease rather abruptly across the boundary. However, linear regression suggests a gradual decline. Linear regression attempts to fit a straight line across what may be a sharp, downward step-like pattern, and thus may not have been the most appropriate analysis for the target fish as a combined group. However, a sharp declining pattern may also result if the level of fishing outside this reserve boundary is high enough to prevent the development of a gradual pattern of decreasing abundance. That is, spillover from the reserve may be present but fishers could deplete rapidly any increase in abundance just outside this boundary. Data from fisheries monitoring (Chapter 4 of this thesis), however, indicate that fishers seem to avoid fishing very close to the boundaries (within 100 m of the boundaries) of Apo Reserve.

Decreasing gradients of abundance across the northern boundary of Apo Reserve were more apparent when targeted fish species were grouped according to similar mobility characteristics. Patterns of gradual decrease in abundance were clearer in the density of sedentary targets, and in the density and biomass of vagile targets (Figures 2.5, 2.6). The spatial patterns of abundance of sedentary and vagile targeted fish (but not highly vagile targeted fish) across the boundary were consistent with predictions of spillover (Rakitin & Kramer 1996). That is, sedentary fish exhibited a steeper pattern of decrease than vagile fish. Rates of decrease in abundance were 2-3 times higher for sedentary than for vagile fish (Figure 2.6). Habitat did not show any significant trends across the northern boundary of Apo Reserve. Furthermore, the results suggest that protection, rather than habitat, were more important in explaining the spatial patterns of abundance of sedentary and vagile targeted fish across this reserve. The gradients of decreasing abundance for sedentary and vagile targeted fish across the northern boundary of Apo Reserve may be due to spillover.

An abrupt decreasing trend of abundance of highly vagile target fish was found across the northern reserve boundary of Apo. This pattern was not predicted for this mobility group. It is unlikely that spillover of highly vagile target fish is present across this boundary of Apo Reserve or that the level of fishing for this group outside this boundary is high enough to prevent the formation of a gradual decreasing pattern. Local fishers fish for many of the species that comprise this guild (*Naso* spp. and Carangidae) mainly at the southernmost and northernmost ends of Apo Island (Bellwood 1988, Chapter 4 of this thesis, R. Bantaya, and G. Mendez, local fishers at Apo Island, *personal communication*). A more plausible reason for the abrupt pattern of decrease in this group is that some species may have been classified incorrectly as ‘highly vagile’. Intensive observation of one species in this group (Chapter 3 of this thesis) suggests that this may partly be the case. Large (>40 cm TL) *Naso vlamingii*, mostly in the reserve, seem to defend territories of not more than 30 m in radius. Alternatively, some mobile species (e.g. Carangidae) may prefer areas in the reserve as resting or feeding sites. Sampling may have occurred, by chance, at times when such fish were resting or feeding in the reserve.

No decreasing trends in the abundance of targeted fish were detected across the southern boundary of Apo Reserve, or the boundaries of Balicasag Reserve. No gradients were detected across control boundaries at either island. The lack of a gradient of abundance across the southern boundary of Apo Reserve over the short distance investigated in this study (about 200 m) is consistent with results of previous studies that suggest that density and biomass of targeted fish (large predators and planktivores) only begin to show a decline beyond 250 m from this boundary (Russ & Alcala 1996a, Russ et al. 2003, 2004). It is likely that the present study would have detected a similar decrease in abundance if sampling in this study had been extended beyond 250 m from the southern reserve boundary. The low density and biomass of targeted fish at the southern portion of the control at Apo Island (Figures 2.2 and 2.3) supports this suggestion. Previous studies have argued that the declining pattern of fish abundance beyond 250 m from the southern boundary of Apo Reserve may have resulted from spillover, which could have begun to be detected by visual census about 8 years after the reserve was protected (Russ & Alcala 1996a, Russ et al. 2003, 2004). In the present study, sampling was not extended beyond 250 m from the southern boundary for safety reasons. Currents are unpredictable and usually strong at the southern end of Apo Island.

The absence of a significant difference in the abundance of targeted fish between the reserve and fished areas at Balicasag, and thus the lack of gradients of abundance across reserve boundaries, may be due to a general increase in abundance of target fish stocks outside the reserve. Data taken before and more than 15 years after the reserve was established at Balicasag suggest that abundance of target fish has increased substantially both inside and outside Balicasag Reserve (G. Russ *unpublished data* 1983, 1999-2000). The reasons for this are not clear. Successful recruitment inside and outside the reserve, and reduced fishing by locals around the whole of Balicasag Island, are possible causes. No data are available to support the hypothesis of successful recruitment. However, information suggests that locals may be fishing less for reef fish around Balicasag Island because of higher income from collecting deep-water shells (B. Stockwell, *unpublished data*). It is unlikely that spillover from the reserve increased stock abundance throughout the whole fishing area at Balicasag Island, simply because reef fishes have limited movement potential, usually on the scale of hundreds of metres (Zeller 1997, Zeller & Russ 1998, Chapman & Kramer 2000, Meyer et al., 2000).

Few studies have demonstrated the presence of gradients of decreasing abundance across reserve boundaries. Fewer studies take account of habitat effects on such gradients. The findings of the present study at Apo Reserve are consistent with those of McClanahan and Mangi (2000) at the no-take Mombasa Marine Park in Kenya. They found decreasing fish abundance and trap catch rate with increasing distance from the two lateral boundaries of the park. They attributed this pattern to spillover from the reserve, since habitat did not vary significantly with distance from the boundaries. Apo Reserve and Mombasa Marine Park are no-take reserves that differ in terms of geographical setting (island reef vs. coastal reef), size (hundreds of meters vs. several kilometres wide), and dominant substratum type (live coral vs. seagrass and coral rubble). If the gradients of decreasing abundance away from the two reserves are due to spillover, movement potential of fish and availability of continuous habitat suitable for reef fish adjacent to reserves may have been important in promoting the net emigration of some fish from the reserves. At Apo Reserve, expanses of habitat that are avoided by many reef fishes, such as sand, may not be frequent enough to inhibit movement of targeted fish from the reserve. In contrast, at the Barbados Marine Reserve, reefs are separated by broad expanses of sand (Rakitin & Kramer 1996). Rakitin and Kramer (1996) found a significant negative relationship between trap catch rate and distance from the centre of this reserve up to 2 km from the reserve boundaries. However, visual census did not detect a significant relationship between density of trappable fish and distance from the reserve centre, despite a higher density of trappable fish inside the reserve. Chapman and Kramer (1999) investigated these contradictory patterns. After statistically controlling for the effects of habitat, no significant negative relationship between fish density and distance away from the reserve was found. However, fish density was significantly higher inside than outside the reserve (Chapman & Kramer 1999). They reasoned that emigration of fish from the reserve was minimal, maintaining higher fish density inside the reserve. This conclusion was supported by an extensive tagging study, which demonstrated that movement of fish between reefs separated by sand was rare (Chapman & Kramer 2000). No movement from the reserve to the fished area was recorded (Chapman & Kramer 2000).

In conclusion, this study detected gradients of decreasing abundance of targeted fish across only the northern boundary of Apo Reserve. Habitat factors could not account for this result. Gradients of decreasing abundance of targeted fish, particularly sedentary and vagile guilds, were consistent with predictions of spillover. However, this study has not demonstrated unequivocally that net fish emigration from the reserve produced this gradient across the northern boundary of Apo Reserve. In fact, no study has found direct evidence for density dependent emigration or short-term movements of fishes within a home range that have resulted in such patterns (see Figure 2.1). To demonstrate unequivocally that such gradients result from spillover, changes in the movement, distribution, and abundance of fish across reserve boundaries need to be monitored before and after reserve establishment (Russ 2002). Nonetheless, four lines of evidence in support of spillover have been documented so far at Apo Reserve. Firstly, regular monitoring over two decades of protection has shown that the abundance of fish targeted by fisheries has increased substantially in the reserve (Russ & Alcala 1996b, Russ et al. 2003, 2004). Secondly, the abundance of some targeted fish has increased over time just outside the southern boundary as protection continued, resulting in a pattern of decreasing abundance away from the boundary (Russ & Alcala 1996a, Russ et al. 2003, 2004). Thirdly, surveys at one point in time have demonstrated gradients of decreasing abundance of targeted fish across the northern reserve boundary that are consistent with spillover (this study). Fourthly, catch per unit effort of some targeted fishes are greater nearer than farther away from the lateral boundaries of the reserve (Russ et al. 2003, 2004). All of these results suggest that spillover of fishes from Apo Reserve may influence fished areas very near to the reserve, but are slow to develop.

Chapter 3

Mechanisms inducing movement of adult fish from a marine reserve

Abstract. Evidence for net export of adult fish from reserves, or spillover, has increased in recent years but the underlying mechanisms involved in this process remain poorly understood. This study provides evidence consistent with two mechanisms for the spillover of a planktivorous reef fish, *Naso vlamingii*, from a small no-take reserve (protected for 20 years) at Apo Island, in the Philippines. The first mechanism may result in temporary movement of adult fish from reserve to fished areas in the course of daily excursions within their home ranges. Underwater visual censuses made twice over several days across a transect that traversed Apo Reserve and the fished areas adjacent to the reserve suggested that some adults of *N. vlamingii* [> 26 cm total length (TL)] could emigrate occasionally from the reserve. Larger adults (41-45 cm TL) may have more limited movement and tend to stay inside the reserve. However, movement of adult *N. vlamingii* across the boundaries of Apo Reserve was seen rarely (only 5% of individuals > 26 cm TL) from direct observations (? 15 min duration). Results from visual censuses and experimental hook and line fishing carried out at different distances from the reserve boundary suggest that the movement of adult *N. vlamingii* from Apo Reserve over the short term (days) may be related to migration to two preferred feeding sites outside the reserve, about 150-200 m away from each lateral boundary of the reserve. The second mechanism involves competition among adult *N. vlamingii* inside Apo Reserve, which may have resulted in home range relocation of smaller adults away from the reserve over the long term. Observations of the behaviour of *N. vlamingii* showed that aggressive interactions among adults were more frequent inside than outside Apo Reserve. This suggests that density-dependent interactions are more intense inside than outside the reserve. When interacting adults differed in their sizes, the larger individual was always observed to chase away the smaller one. Inside the reserve, fleeing from a larger conspecific was observed only in the 31-35 cm TL size class, and only in the 26-30 cm TL size class outside the reserve. These sizes were similar to the size range of adults caught outside the reserve during experimental hook and line fishing (25.6 to 38 cm TL). Furthermore, the size of adult fish captured by

experimental fishing decreased as one moved away from the reserve boundaries (from 50-100 to 250-300 m). This pattern is consistent with density-dependent home range relocation of fish from the reserve. The evidence for the movement of adult fish from the reserve over the short-term (days) is equivocal. Direct approaches (e.g. tagging) to study movement of adult fish across the boundaries of Apo Reserve were impractical because of social factors. The evidence consistent with density-dependent movement of smaller adult fish driven by competition inside the reserve, on its own, is also equivocal. Nevertheless, data collected independently at Apo Reserve from 1983 to 2003 show that density and mean size of *N. vlamingii* have increased substantially in the reserve and in an area just outside the boundary of the reserve over time. The patterns found here that are consistent with a density-dependent mechanism of spillover may have developed over 20 years of reserve protection.

3.1 Introduction

Net export of adult fish from reserves, or spillover is often assumed to be driven by density-dependent effects (Roberts & Polunin 1991, Sanchez Lizaso et al. 2000, Russ 2002). If protection from fishing increases the density of fish inside a reserve, effects such as competition for resources among fish may intensify in the reserve. This may result in some individuals relocating their home ranges outside the reserve (Kramer & Chapman 1999). However, spillover through density-dependent movement of adult fish may take years or decades to develop (Russ 2002) because recovery of populations in reserves will depend on many factors such as initial population size, intrinsic rate of population growth, success of recruitment, flux rates across reserve boundaries, and the degree of reduction of fishing mortality inside the reserve (Jennings 2001). On the other hand, it is also often assumed that spillover may occur through the occasional movement of some fish from a reserve (Roberts & Polunin 1991, Gell & Roberts 2002, Russ 2002). Fish may move from a reserve during daily or seasonal movements within their home ranges (Holland et al. 1993, Holland et al. 1996, Zeller & Russ 1998, Meyer et al. 2000, Eristhee & Oxenford 2001). Also, larger fish may have higher potential for movement because they have larger home ranges (Kramer & Chapman 1999). High movement rates of individuals across reserve boundaries may slow down the recovery rate of populations in reserves (Jennings 2001). Recovery of populations in reserves, and subsequent spillover, would require that some individuals residing in a reserve usually remain within the reserve (Willis et al. 2001). Reserves that are established for the benefit of local fisheries must somehow be large enough to promote population recovery, yet small enough to permit some spillover.

The use of no-take reserves for the purpose of enhancing adjacent fisheries remains controversial (Russ 2002, Gell & Roberts 2003). So far, some of the best evidence for spillover comes from monitoring of reserves, which have shown increased abundance of targeted fish inside them and in adjacent areas open to fishing over time (McClanahan & Kaunda-Arara 1996, Russ & Alcala 1996, McClanahan & Mangi 2000, Roberts et al. 2001, Russ et al. 2003, 2004). However, none of these studies have provided evidence for mechanisms inducing movement of adult fish from reserve to fished areas. Evidence for such mechanisms is crucial in demonstrating that reserves benefit adjacent fisheries through the export of adult fish. However, direct evidence for density-dependent mechanisms and/or occasional movement of adult fish developing in no-take reserves over time may be difficult to obtain (Russ 2002). Few have attempted to investigate these processes experimentally (Zeller et al. 2003).

Competition for resources (usually food, shelter or breeding partners) in coral reef fishes is often manifested in aggressive interactions (interference competition) among individuals (Shulman 1985, Robertson & Gaines 1986, Clifton 1990, Mumby & Wabnitz 2002). A higher frequency of aggressive interactions inside than outside a reserve may develop over time if increased population density results in greater competition for resources inside a reserve. This may indicate the potential for density-dependent net emigration from the reserve. However, the effect of competition on movement of reef fishes is poorly understood (Jones 1991, Jones & McCormick 2002). If larger body size of reef fish imparts superiority in competition (Robertson 1998), frequent aggressive interactions among individuals inside the reserve may lead eventually to larger individuals driving out smaller individuals from the reserve. This may result in a pattern of higher to lower mean size of individuals from inside to outside the reserve (Rakitin & Kramer 1996, Kramer & Chapman 1999). Surprisingly, no studies have examined how the intensity of competitive interactions differs inside and outside no-take reserves. Also, few studies have verified the predicted pattern of decreasing mean size from inside to outside a reserve.

Home range relocation is considered rare in reef fishes because many species display strong site-attachment (Sale 1978). Fishes of smaller body size, in particular, may be reluctant to relocate because of a higher cost of growth and/or risk of mortality associated with relocation (Kramer & Chapman 1999). Nonetheless, evidence for home range relocation in reef fishes is increasing (Robertson 1988, Lewis 1997). Furthermore, larger or more mobile reef fishes may display movement that is significant at presumed spatial scales of spillover (tens of metres to a few kilometres). Some reef fishes, for instance, may migrate across different habitats, reef zones, or along a depth gradient during ontogeny (Meyer et al. 2000, Cocheret de la Moriniere et al. 2002, Nagelkerken & van der Velde 2002). Ontogenetic movement may indicate behavioural decisions that aim to maximise net benefits by increasing growth rate and lowering risk of mortality (Dahlgren & Eggleston 2000). It is likely that the behavioural decisions leading to shifts in habitat are influenced by density-dependent factors.

This study provides evidence consistent with two mechanisms for the spillover of a surgeonfish (Acanthuridae), *Naso vlamingii*, from a small no-take reserve at Apo Island, in the Philippines. *N. vlamingii* is one of the several larger species of surgeonfish that are exploited by the local fishery at Apo Island. The species prefers seaward reef slopes, from 4 to 50 m deep (Lieske & Myers 1997). Individuals can grow up to 50 cm TL (Lieske & Myers 1997) and may live as long as 45 years (Choat & Axe 1996). Adults feed on gelatinous zooplankton (ctenophores) during the day (Lieske & Myers 1997). There are no available studies on the social organization of *Naso vlamingii*, but adults are usually observed solitary or in pairs and may form aggregations off steep reef slopes and drop-offs when feeding. Larger individuals (presumably males) have the ability to display their blue body markings during interactions with conspecifics or other species. Such a display may be related to territorial behaviour during feeding or mating. Territorial systems and dominance hierarchies appear to be common in surgeonfishes (Jones 1968, Robertson et al. 1979, Roberston & Gaines 1986, Montgomery et al. 1989, Craig 1996). A recent study by Morgan and Kramer (2004) showed that the incidence of territoriality in a Carribean surgeonfish, *Acanthurus coeruleus*, increased with increasing density of conspecifics.

The present study builds on previous work on *N. vlamingii* at Apo Island by Russ et al. (2003). They documented an increase in the abundance of the species in the reserve and an adjacent fished area, over almost two decades (Russ et al. 2003). They also found higher catch rates of *N. vlamingii* closer to than further away from the boundaries of the reserve (Russ et al. 2003). In the present study, the short-term (within day/s) movement of *N. vlamingii* from reserve to fished areas was investigated. Movement was assessed indirectly from visual censuses made twice across a transect traversing the reserve and adjacent fished areas, and directly from visual tracking of individual *N. vlamingii*. The spatial pattern of catch per unit effort and mean size of *N. vlamingii* outside the reserve was determined by experimental hook and line fishing. Lastly, the frequencies of aggressive interactions among different size classes of *N. vlamingii* were investigated to determine if density-dependent interactions were more frequent inside than outside the reserve.

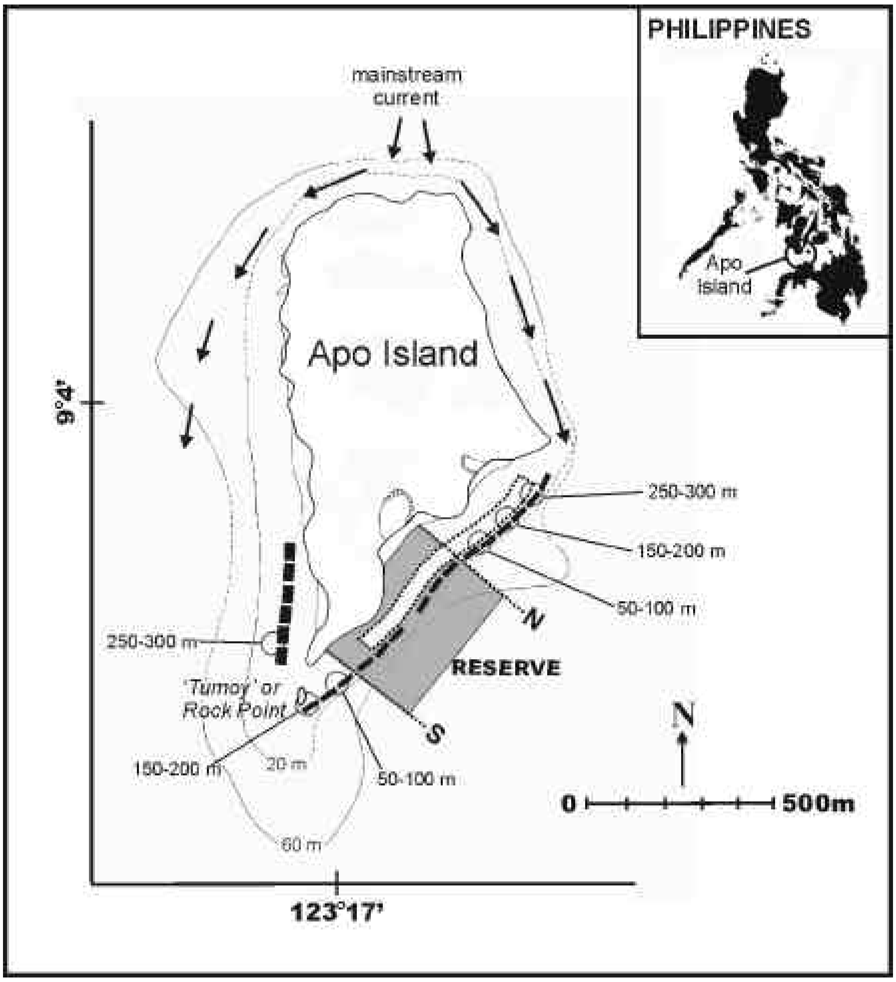


Figure 3.1. Map of Apo Island showing the no-take reserve (shaded). Black rectangles traversing the reserve show the approximate positions of individual 50 x 10 m transects where underwater visual censuses were made. Circles indicate the approximate positions of sites outside the reserve that were fished for *Naso vlamingii* using traditional hook and line. The distances of these sites from the northern (N) or southern boundary (S) are also shown. The white rectangle enclosed by broken lines shows the approximate area where visual observations of the movement and behaviour of individual *N. vlamingii* were made. Black rectangles on the southwestern side of Apo Island show the approximate positions of transects in the non-reserve site surveyed by G. Russ from 1983 to 2003 (see Figures 3.6 and 3.7).

3.2 Materials and Methods

3.2.1 Study site

Apo Island (9? 4’ N, 123? 17’ E) is located in the central Philippines, southeast of Negros Island (Figure 3.1). It is a small volcanic island inhabited by a community of about 700 residents. The island has a coral reef area of about 0.54 km^2 (to the 20 m isobath). Reef fisheries and tourism are the main sources of income for the community (Russ & Alcala 1999). A 450 m long no-take reserve on the southeastern side of Apo Island (Figure 3.1) was established in 1982 (Russ & Alcala 1999). Protection of the reserve from fishing is strictly enforced by the community, with good compliance from local people and visitors (Russ & Alcala 1999). Outside the reserve, fishers from Apo Island and from nearby towns in Negros Island engage in artisanal fishing using methods non-destructive to coral habitats (Chapter 4). The coral reef slope in the reserve and in fished areas close to the lateral boundaries of the reserve is relatively steep and rugose, with high hard coral cover (Chapter 2). Apo Island is influenced by a strong mainstream northerly current (Figure 3.1). This makes the northern side of the island favourable for fishing, targeting mostly Carangidae (jacks) during the SW monsoon (June to September) and in the interim calm periods (April, May and October) (Bellwood 1988, Chapter 4). However, during the NE monsoon (November to March), locals tend to fish the southern and western sides of the island (Chapter 4). Local people consider the southern end of Apo Island as a traditional fishing ground for *Naso vlamingii* (called ‘bongkokan’ in the local dialect), referring to this area as *Tumoy* or Rock Point (M. Pascobello, Apo Island resident/Barangay Chairman, *personal communication*) (Figure 3.1). Fishers at Apo Island catch *N. vlamingii* using large bamboo traps, spears and hook and line.

3.2.2 Visual census in the reserve and fished areas adjacent to the reserve

The visual census data are the same as used in Chapter 2. The sampling program for these censuses was originally designed to investigate if gradients of abundance of targeted fish were present across the reserve boundaries. Fifteen 50 m x 10 m transects were surveyed across a large transect traversing Apo Reserve and the two fished areas adjacent to the lateral boundaries of the reserve, up to 200-400 m beyond the reserve boundaries (Figure 3.1). The transects were placed sequentially (at least 10 m apart) parallel to the shore, along the reef slope at a depth of 10-12 m. Fish surveys were made along these transects by one diver using SCUBA (R.A.A.). Ninety-five species of targeted fish were censused. All targeted fish >10 cm total length (TL) were recorded and their size (cm TL) estimated. Before surveys, size estimation was calibrated by estimating the lengths of PVC pipes of 35 different sizes underwater (size range: 10-78 cm). All 50 m x 10 m transects were surveyed twice. However, all transects were surveyed once before they were resurveyed. The first set of surveys was made from November 22-23, 2002, and the second set was made from November 25-29, 2002. Three neighbouring 50 m x 10 m transects were surveyed per dive. The order of dives amongst transects was random (weather permitting). It was possible to place transects within 5-10 m of their first sampling position by making use of permanent mooring buoys, natural underwater features and a GPS receiver. All survey dives were made between 0700-1600 hours. The density of *Naso vlamingii* on each transect was expressed as number per 500 m^2. A repeated-measures ANOVA was used to examine how *N. vlamingii* density varied among the three sites (reserve, adjacent fished area north of the reserve, adjacent fished area south of the reserve) over the two sampling times. A 2-way ANOVA was used to examine if mean size of *N. vlamingii* varied among sites and sampling times. Multiple comparisons of mean sizes among sites were made using Tukey’s test (Zar 1999).

3.2.3 Experimental fishing adjacent to the reserve

Two experienced local fishers conducted traditional hook and line fishing in the two fished areas adjacent to the lateral boundaries of Apo Reserve (Figure 3.1). The hook and line technique they used specifically targeted *Naso vlamingii* (but other planktivorous and some omnivorous fishes were also caught). This technique involves using a very small hook attached to a fine line made of nylon and twine. The hook is baited with a small piece of fish skin or intestine (usually from *Decapterus* spp.). Using the twine portion of the line, the hook is wrapped around a small stone together with finely minced fish chum. The line is secured around the stone using a type of slip-knot. At the desired depth, the line is pulled to untie the knot, release the stone, scatter the chum and free the hook. Planktivorous fish are attracted to the scattered chum and usually reach the baited hook first. Hook and line fishing for *N. vlamingii* was done above the reef slope (between 10 and 20 m deep) in the two fished areas, at three distances from the reserve boundary: 50-100, 150-200 and 250-300 m (Figure 3.1). Marker buoys were installed at each distance from the reserve with the aid of a GPS. Each distance was fished during both the flood and ebb tides. Eight replicates of one hour of fishing were made at each distance for each tide (96 hours of experimental fishing for the whole study). Experimental fishing was done from September 20 to October 3, 2003, between 0700-1700 hours. Fishing amongst sites was random (weather permitting). All *N. vlamingii* caught were weighed using a market scale accurate to 0.01 kg and their lengths (cm TL) measured. Catch rates of *N. vlamingii* were expressed in fish person^{-1} hr^{-1} and kg person^{-1} hr^{-1}. A 3-way ANOVA was used to examine how catch rates of *N. vlamingii* varied among the two fished areas, three distances from the reserve boundaries, and two tides. Multiple comparisons of catch rates at each distance category were made using Tukey's test. A 2-way ANOVA was used to examine how mean size of *N. vlamingii* varied among the fished areas and distances from the reserve boundary.

3.2.4 Observation of movement and aggressive interactions

One person on snorkel (R.A.A.) observed the movement and behaviour of individual *Naso vlamingii* in the reserve and in the fished area adjacent to the northern boundary of the reserve. Observations were not made in the fished area adjacent to the southern boundary of the reserve for safety reasons (currents in this area are unpredictable and usually strong). Observations were made on the reef slope, in an area about 600 m long between the 5 and 12 m isobaths (Figure 3.1). This area was divided equally into six sectors, three sectors each in the reserve and in the fished area. Four observations of individuals were allotted to each of three size classes (?30, 31-40, ?41 cm TL) in each sector. Sampling was made randomly among sectors and among size classes, but neighbouring sectors were sampled at the same time of day (morning, noon or afternoon) in order to decrease the time required to swim among sectors. However, sampling moved on to a different size class (sometimes this size class was in a different sector) if no individual of a given size class was seen for at least 15 min within a sector.

At the beginning of each observation period, the observer estimated the length (cm TL) of the individual and noted its starting position. The observer made use of a conspicuous natural underwater feature to identify the starting position. The observer also estimated the distance of the starting position from the northern boundary of the reserve using landmarks on shore. The maximum distance moved laterally by the individual towards the northern or southern reserve boundary, in increments of 10 m, was recorded every minute. Behavioural interactions with conspecifics were also recorded every minute. The behaviours observed were: aggression towards smaller conspecifics, aggression towards larger conspecifics, aggression towards similarly-sized conspecifics, fleeing from a smaller conspecific, fleeing from a larger conspecific, and fleeing from a similarly-sized conspecific. Other behaviours (e.g. feeding) and aggressive interactions with other species were also recorded. Individuals were considered to show aggressive behaviour when they chased another individual away. Larger individuals also show aggressive behaviour by displaying blue body markings to another individual. However, larger individuals may also display their blue markings without also showing aggressive behaviour, such as when visiting a cleaning station. Individuals were observed for a maximum of 15 min or until they were lost from view (observation time for each individual in reserve: range = 3-15 min, mode = 15 min; fished area: range = 3-15 min, mode = 10 min). Only individuals that were observed for a minimum of 4 min were included in the analyses of movement and behaviour. For each individual, the frequency of an observed behaviour was estimated as the number of times the behaviour was observed, divided by total observation time (min). The observer always kept a reasonable distance (about 5 m) from the fish to avoid influencing its natural behaviour. All observations were made from December 5 to 8, 2003, between 0700-1600 hours. The observer spent approximately 35 hours snorkelling.

3.3 Results

3.3.1 Changes in density and size distribution in the reserve and adjacent fished areas in the short-term

The spatial pattern of abundance of *Naso vlamingii* in Apo Reserve and the two adjacent fished areas changed considerably in a short period (several days). During the first sampling (22-23 November, 2002), the mean density of *N. vlamingii* was higher in the reserve than in the two fished areas [reserve: 6.0 ? 1.3 (SE) fish 500 m^{-2}, fished area north of the reserve: 1.8 ? 1.3 fish 500 m^{-2}, fished area south of the reserve: 2.7 ? 2.2

fish 500 m^{-2}, Figure 3.2a]. However, during the second sampling (25-29 November, 2002), the mean density of *N. vlamingii* was higher in the two fished areas than in the reserve (reserve: 2.3 ? 1.0 fish 500 m^{-2}, fished area north of the reserve: 3.2 ? 1.3 fish 500 m^{-2}, fished area south of the reserve: 7.3 ? 1.9 fish 500 m^{-2}, Figure 3.2b). The change in mean density was greatest in the fished area adjacent to the southern boundary of the reserve (a 170% increase). However, during the two sampling times, the density of *N. vlamingii* was always high in the fished areas approximately 200 m away from the northern boundary (8 fish per 500 m^2 in the two sampling times), and approximately 150 m away from the southern boundary of the reserve (7 and 11 fish per 500 m^2 in the first and second sampling, respectively) (Figure 3.2a, b). The highest density recorded in the reserve was 10 fish per 500 m^2. Neither site nor sampling time had a significant effect on the density of *N. vlamingii* over the sampling period (site: $F_{2, 12} = 1.04$, $p = 0.38$; sampling time: $F_{1, 12} = 1.41$, $p = 0.26$). However, the site x sampling time interaction was highly significant ($F_{2, 12} = 13.56$, $p = 0.001$).

Site and sampling time had significant effects on mean size of *Naso vlamingii* (site: $F_{2, 104} = 31.07$, $p < 0.001$; sampling time: $F_{1, 104} = 36.96$, $p < 0.001$). However, the site x sampling time interaction was significant ($F_{2, 104} = 7.93$, $p = 0.001$). During the first sampling time, mean size of *N. vlamingii* in the reserve (43.1 ? 1.0 cm TL) was significantly larger than in the fished area south of the reserve (27.0 ? 1.6 cm TL) (Tukey's test: $q_{3, 104} = 10.9$, $p < 0.001$) and the fished area north of the reserve (37.0 ? 1.2 cm TL) ($q_{3, 104} = 4.7$, $p < 0.005$) (Figure 3.2a). During the second sampling time, mean size in the reserve (46.1 ? 0.8 cm) was also significantly larger than in the fished area south of the reserve (40.6 ? 1.4 cm) ($q_{3, 104} = 4.3$, $p < 0.01$) and the fished area north of the reserve (41.2 ? 0.9 cm) ($q_{3, 104} = 3.7$, $p < 0.05$) (Figure 3.2b). However, mean size did not differ between the two fished areas during both sampling times (first sampling time: $q_{3, 104} = 1.7$; second sampling time: $q_{3, 104} = 0.5$). Mean size in the fished area south of the reserve was significantly higher during the second compared to the first sampling time ($q_{2, 104} = 8.7$, $p < 0.001$). The same was true for mean size in the fished area north of the reserve ($q_{2, 104} = 2.9$, $p < 0.05$). The increase in mean size from the first to the second sampling for the fished area south of the reserve was 50%, for the fished area north of the reserve, 11%. However, mean size inside Apo Reserve did not significantly differ from the first to the second sampling time ($q_{2, 104} = 2.5$, $p > 0.05$).

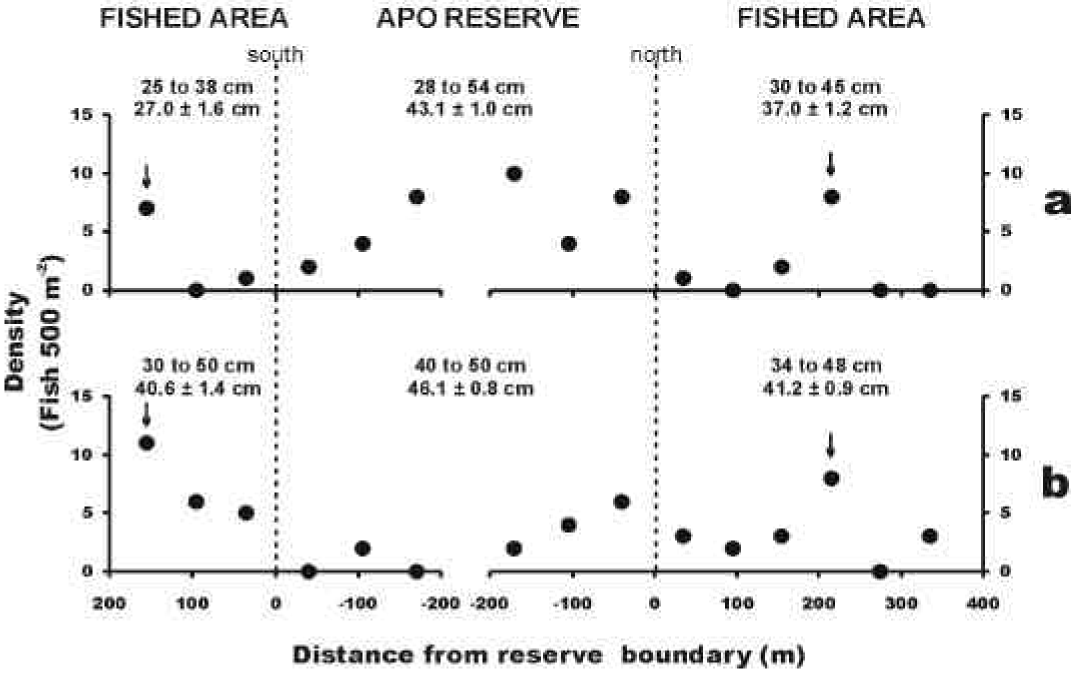
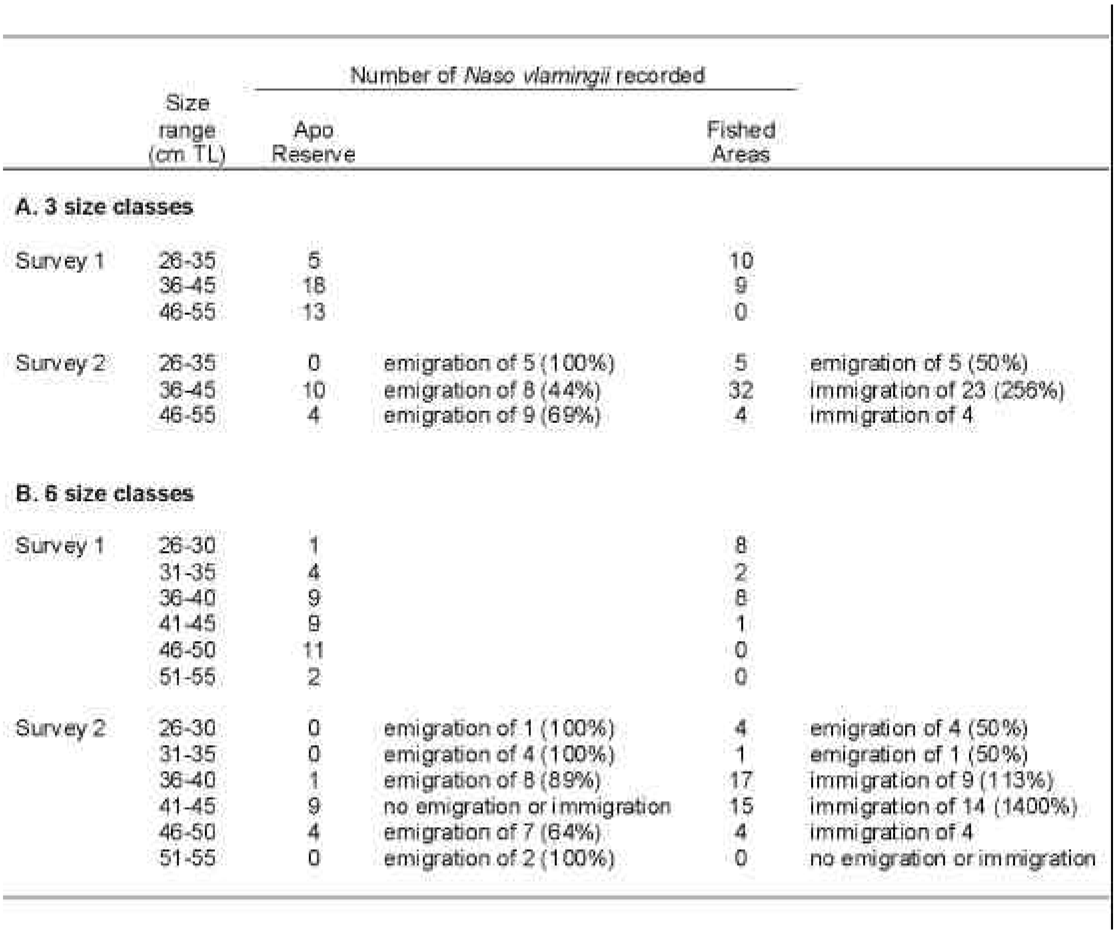


Figure 3.2. The mean density of *Naso vlamingii* across Apo Reserve and the two adjacent fished areas outside the reserve estimated from underwater visual census. (a) First survey (November 22-23, 2002); (b) Second survey (November 25-29, 2002). Survey depths on the reef slope were 10-12 m in the reserve and non-reserve. The northern and southern boundaries of Apo Reserve are indicated by broken vertical lines. Arrows indicate where mean density remained high over the two surveys (about 150-200 m away from the reserve boundaries; see also the results of experimental hook and line fishing, Figure 3.3). The size range (upper numbers, cm TL) and mean size (lower numbers, cm TL) of *N. vlamingii* recorded in the reserve and the two adjacent fished areas are shown for each

Fewer individuals in the smallest of three size classes (26-35 cm TL) were recorded during the second sampling across the three sites (Table 3.1 A.). For the reserve, fewer individuals in the larger size classes (36-45 and 46-55 cm TL) were recorded during the second sampling (Table 3.1 A.). However, for the fished areas, more individuals in larger size classes (36-45 and 46-55 cm TL) were recorded during the second sampling (Table 3.1 A.). The size frequency distribution was also examined based on six size classes (Table 3.1 B.). Smaller individuals (26-30, 31-35 cm TL) were less abundant in the reserve and the fished areas during the second sampling. Larger individuals (36-40, 46-50, 51-55 cm TL) were less abundant in the reserve during the second sampling, but larger individuals (36-40, 41-45, 46-50 cm TL) were more abundant in the fished areas during the second sampling. However, for the reserve, no change was observed in the frequency of individuals in the 41-45 cm TL size class (Table 3.1 B.). Also, no individuals in the 51-55 cm TL size class were observed in the fished areas at any time (Table 3.1 B.).

Table 3.1. Changes in the size frequency distribution of *Naso vlamingii* in Apo Reserve and the fished areas adjacent to the reserve from the first survey (November 22-23, 2002) to the second survey (November 25-29, 2002) using visual census. Data were divided into three size classes (A.) and six size classes (B.). Changes in the number of individuals recorded for each size class were interpreted as emigration from or immigration to the reserve or fished areas. The percent change in the numbers recorded from the first to the second survey is enclosed in parentheses.



	Size range (cm TL)	Number of *Naso vlamingii* recorded			
		Apo Reserve		Fished Areas	
A. 3 size classes					
Survey 1	26-35	5		10	
	36-45	18		9	
	46-55	13		0	
Survey 2	26-35	0	emigration of 5 (100%)	5	emigration of 5 (50%)
	36-45	10	emigration of 8 (44%)	32	immigration of 23 (256%)
	46-55	4	emigration of 9 (69%)	4	immigration of 4
B. 6 size classes					
Survey 1	26-30	1		8	
	31-35	4		2	
	36-40	9		8	
	41-45	9		1	
	46-50	11		0	
	51-55	2		0	
Survey 2	26-30	0	emigration of 1 (100%)	4	emigration of 4 (50%)
	31-35	0	emigration of 4 (100%)	1	emigration of 1 (50%)
	36-40	1	emigration of 8 (89%)	17	immigration of 9 (113%)
	41-45	9	no emigration or immigration	15	immigration of 14 (1400%)
	46-50	4	emigration of 7 (64%)	4	immigration of 4
	51-55	0	emigration of 2 (100%)	0	no emigration or immigration

3.3.2 *Catch rates and size distribution of individuals caught outside the reserve*

Catch per unit effort (CPUE) of *Naso vlamingii* (in fish person^{-1} hr^{-1} and in kg person^{-1} hr^{-1}) was highest at 150-200 m outside the boundaries of Apo Reserve (Figure 3.3a, b). CPUE differed significantly with distance from reserve boundaries ($F_{2, 84}$ = 7.10, p = 0.001, for fish person^{-1} hr^{-1}; $F_{2, 84}$ = 6.91, p = 0.002, for kg person^{-1} hr^{-1}), but not between tides ($F_{1, 84}$ = 0.51, p = 0.48 for fish person^{-1} hr^{-1}; $F_{1, 84}$ = 0.41, p = 0.53 for kg person^{-1} hr^{-1}) nor between the two fished areas (north and south) adjacent to reserve boundaries ($F_{1, 84}$ = 1.15, p = 0.29 for fish person^{-1} hr^{-1}; $F_{1, 84}$ = 0.16, p = 0.69 for kg person^{-1} hr^{-1}). There was no significant interaction among distances, tides and fished areas. CPUE at 150-200 m was significantly higher than at 50-100 m and 250-300 m

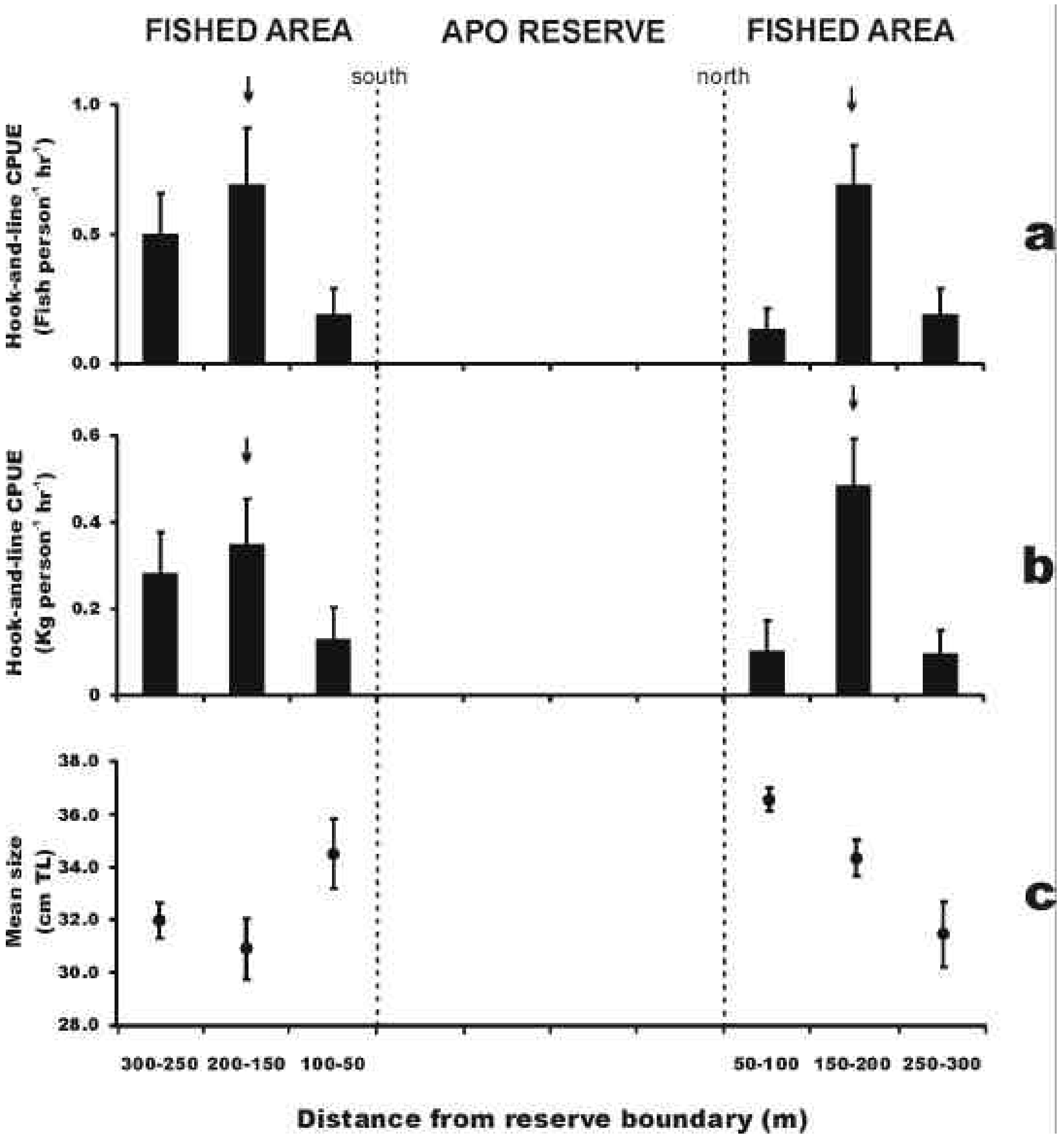


Figure 3.3. Results of experimental hook and line fishing for *Naso vlamingii* outside Apo Reserve at three distances (50-100, 150-200, 250-300 m) from the reserve boundaries. Fishing was done above the reef slope between 10-20 m. The northern and southern boundaries of Apo Reserve are indicated by broken vertical lines. (a) Catch per unit effort (CPUE) in fish person^{-1} hr^{-1}; (b) CPUE in kg person^{-1} hr^{-1}. Arrows indicate where CPUE was highest (about 150-200 m away from the reserve boundaries; see also the results of underwater visual census, Figure 3.2). (c) Mean size of individuals caught at each distance from the reserve boundary. Mean size was highest nearer than further away from the reserve boundaries. Error bars are 1 SE.

(Tukey's test: 150-200 m vs. 50-100 m – $p = 0.001$ for fish person^{-1} hr^{-1}, $p = 0.002$ for kg person^{-1} hr^{-1}; 150-200 m vs. 250-300 m – $p = 0.05$ for fish person^{-1} hr^{-1}, $p = 0.02$ for kg person^{-1} hr^{-1}), but CPUE at 50-100 m and 250-300 m did not differ significantly from each other (Tukey test: $p = 0.39$ for fish person^{-1} hr^{-1}, $p = 0.66$ for kg person^{-1} hr^{-1}). CPUE had a tendency to remain high 250-300 m from the southern boundary of the reserve (Figure 3.3a, b).

Naso vlamingii that were caught outside the reserve were smaller (mean size: 32.8 ? 0.5 cm, size range: 25.6 to 38.0 cm, n = 38) compared to those recorded in visual censuses (size range: 26 to 54, maximum size recorded inside the reserve only). The mean size of fish captured by hook and line was greater closer (50-100 m) to than further away (250-300 m) from the boundaries of Apo Reserve (Figure 3.3c). The fished area adjacent to the northern boundary of the reserve showed a continuous decline of mean size (Figure 3.3c). A 2-way ANOVA did not detect significant differences in mean size between the two fished areas ($F_{2,32} = 2.18$, $p = 0.15$), nor among the three distances from the reserve boundaries ($F_{2,32} = 3.04$, $p = 0.06$).

3.3.3 Movement and behavioural interactions among different size classes

More *N. vlamingii* were seen in the reserve than in the fished area adjacent to the northern boundary of the reserve during behavioural observations (reserve: n = 21, fished area: n = 12). Individuals that were observed in the reserve were larger on average (reserve – mean size: 32.6 ? 2.0 cm, range = 19 to 45 cm; fished area – mean size 23.4 ? 2.2 cm, range = 16 to 38 cm; ANOVA: $F_{1,31} = 8.50$, $p = 0.007$). Larger size classes (31-35, 36-40 and 41-45 cm TL) were more common in the reserve, than in the fished area (Figure 3.4a). No individuals larger than 38 cm TL were seen in the fished area. However no individuals larger than 45 cm TL were seen in the reserve (the largest size recorded in the reserve during underwater visual census was 54 cm TL). This may be due to the limitations of snorkelling. Larger individuals may tend to stay on the deeper reef slope (> 12 m) more often.

The maximum lateral movement (in one direction) observed from a starting point was 40 m. This was recorded for a 16 cm TL individual during a 15 min observation period in the fished area. The maximum lateral movement of 90% of individuals ranged from 20 to 30 m from their starting point. Two individuals in the

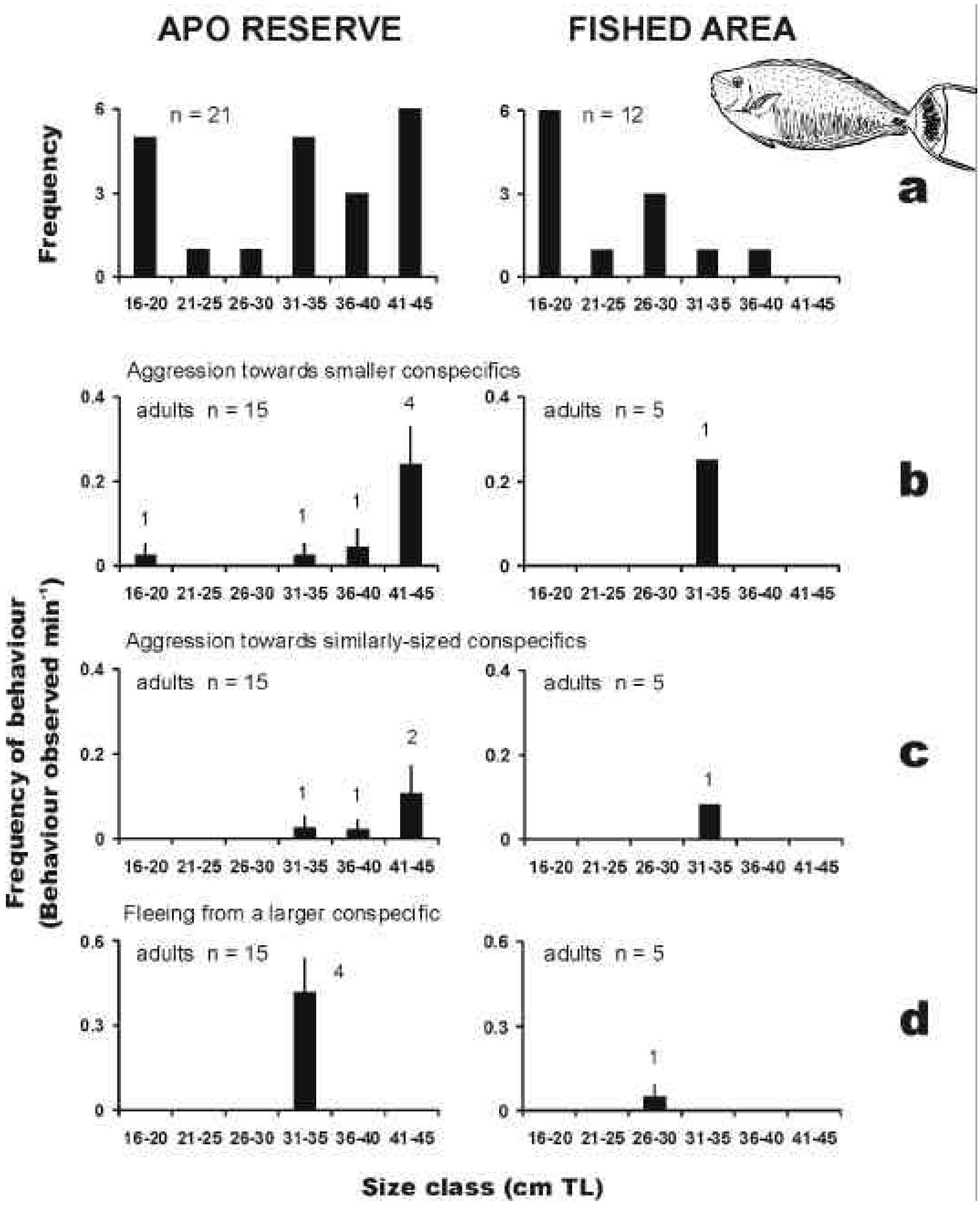


Figure 3.4. Results of observations of the behaviour of *Naso vlamingii* in Apo Reserve and the fished area north of the reserve (refer to Figure 3.2). (a) Size frequency distribution; (b) Frequency of aggression towards smaller conspecifics; (c) Frequency of aggression towards similarly-sized conspecifics; (d) Frequency of fleeing from a larger conspecific. The frequency of a behaviour is averaged for each size class and the number of individuals which displayed that behaviour is indicated. Error bars are 1 SE. Juveniles (16 to 25 cm TL) were not observed to interact with adults (> 26 cm TL). Number of adults observed are shown for (b), (c) and (d). Observations were done on the reef slope between depths of 5 and 12 m.

reserve (38 and 41 cm TL) were never observed to move more than 10 m from their starting point. Maximum lateral movement decreased with increasing size of fish (maximum lateral movement = -0.27 x size + 30.80, $r^2 = 0.16$, $F_{1, 29} = 5.70$, $p = 0.02$; Figure 3.5a). However, maximum lateral movement increased with longer observation time (maximum lateral movement = 0.57 x observation time + 17.58 $r^2 = 0.16$, $F_{1, 29} = 5.41$, $p = 0.03$; Figure 3.5b).

Movement across reserve boundaries was observed rarely. Only three individuals (9% of total observations) were seen to cross the northern boundary of the reserve. Two of these fish were only 20 cm TL. They were observed first inside the reserve and were seen to move a distance of about 5 and 10 m into the fished area (duration of observation for both individuals: 15 min). Only one out of 20 larger individuals >26 cm TL (5% of observations) was observed to cross the boundary. This was a 33 cm TL individual that was observed first on the northern boundary and was seen to move approximately 20 m into the fished area (duration of observation: 6 min). No individuals from the fished area were observed to move into the reserve. No individuals from the reserve were observed to cross the southern boundary of the reserve.

Aggressive interactions between smaller individuals < 26 cm TL and larger individuals > 26 cm TL were observed rarely. The former are probably juveniles less than 5 years old (Choat & Robertson 2002). Juveniles were found usually on the shallower reef slope or on the reef flat (3-5 m deep), sometimes in small, loose groups of 3-20 individuals. Some juveniles were observed to graze on algae growing on hard substratum, and feed on plankton. Aggressive interactions among juveniles were rare. Only 1 of 11 juveniles observed displayed aggressive behaviour (chasing away a smaller juvenile). Individuals > 26 cm TL are probably adults between 5 and 40 years old (Choat & Robertson 2002). Adults were found usually on the deeper reef slope (> 5 m deep). Adults were observed to feed on plankton in open water up to > 20 m away from the reef slope. Aggressive interactions among adults were more common. Seventy-one percent of aggressive interactions among adults occurred during feeding.

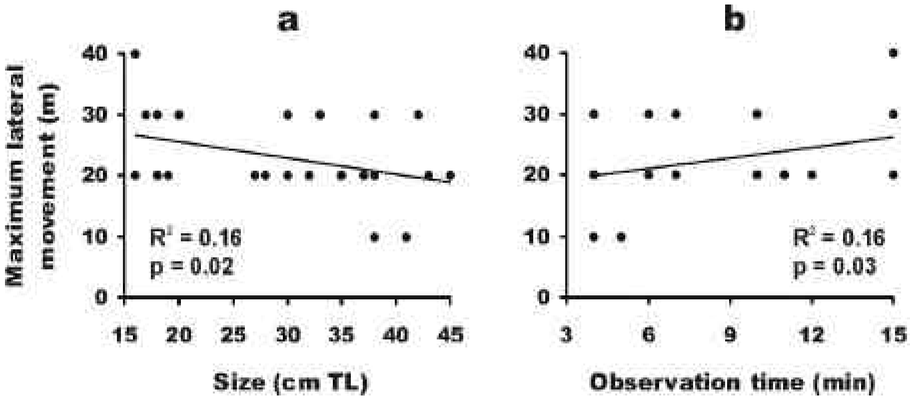


Figure 3.5. Maximum lateral movement (in one direction) versus (a) size, and (b) observation time for individuals of *Naso vlamingii* observed in Apo Reserve and the fished area adjacent to the northern boundary of the reserve.

Aggressive interactions among adults were more frequent in the reserve than in the fished area. In the reserve, 6 of 15 adults (40%) displayed aggressive behaviour towards smaller conspecifics, while 4 of 15 adults (27%) displayed aggressive behaviour towards similarly-sized conspecifics. In contrast, in the fished area, only 1 of 5 adults (20%) displayed aggressive behaviour towards smaller or similarly-sized conspecifics. In the reserve, aggressive behaviour was observed most frequently in adults in the 41-45 cm size class (Figure 3.4b, c). In the fished area, aggressive behaviour was observed only in the 31-35 cm size class (1 individual only) (Figure 3.4b, c). Aggressive interactions between adults of different sizes always resulted in the larger individual chasing away the smaller one. However, when interacting adults were of similar size, no chasing occurred. Fleeing from a larger conspecific was observed more frequently in the reserve than in the fished area. In the reserve, fleeing from a larger conspecific was observed only in the 31-35 cm size class (4 of 15 adults or 26%) (Figure 3.4d). In the fished area, fleeing from a larger conspecific was observed only in the 26-30 cm size class (1 of 5 adults of 20%) (Figure 3.4d). No smaller individual was observed to chase away a larger conspecific. No larger individual was observed to flee from a smaller conspecific. Mean observation time for adult *N. vlamingii* did not differ between the reserve (8.4 ? 1.2 min, n = 15) and the fished area (7.8 ? 1.4 min, n = 5) ($F_{1,18} = 0.07$, $p = 0.80$).

3.4 Discussion

This study investigated if mechanisms of spillover were present in a no-take marine reserve. No other study has investigated the mechanisms involved in the movement of adult fish from reserve to fished areas when spillover has been argued to occur. This study found indirect evidence for the occasional movement of some adult fish from Apo Reserve over the short term. Evidence consistent with a density-dependent mechanism for spillover from the reserve was also found.

The data from underwater visual census suggests that some adults of *Naso vlamingii* (> 26 cm TL) that were inside the reserve initially, could move to fished areas adjacent to the reserve over a short period (days) (Figure 3.2). Both small (26-35 cm TL) and large (36-55 cm TL) adults may have emigrated from the reserve to fished areas (Table 3.1). In particular, larger adults in the 36-40 and 46-50 cm TL size classes may have moved from the reserve to fished areas just outside of the reserve. For example, from the first to the second sampling, eight individuals in the 36-40 cm TL size class possibly emigrated from the reserve, and nine individuals possibly immigrated to the fished adjacent fished areas (Table 3.1 B.). Although the numbers of *N. vlamingii* recorded in the first and second sampling were identical (n = 55), immigrants to the fished areas, especially those in the 41-45 cm TL size class, may have also come from elsewhere besides the reserve (Table 3.1 B.). On the other hand, adults in the 41-45 cm TL size class may have tended to stay inside the reserve (Table 3.1 B.). The finding that movement of *N. vlamingii* may become more limited with increasing size (15 to 45 cm TL, Figure 3.5a) complements this result. Limited movement of some larger adults (approximately 45 cm TL) may have helped to maintain a consistently higher mean size in the reserve than in adjacent fished areas. Mean size inside the reserve did not differ significantly from the first (43.1 ? 1.0 cm TL) to the second sampling (46.1 ? 0.8 cm TL).

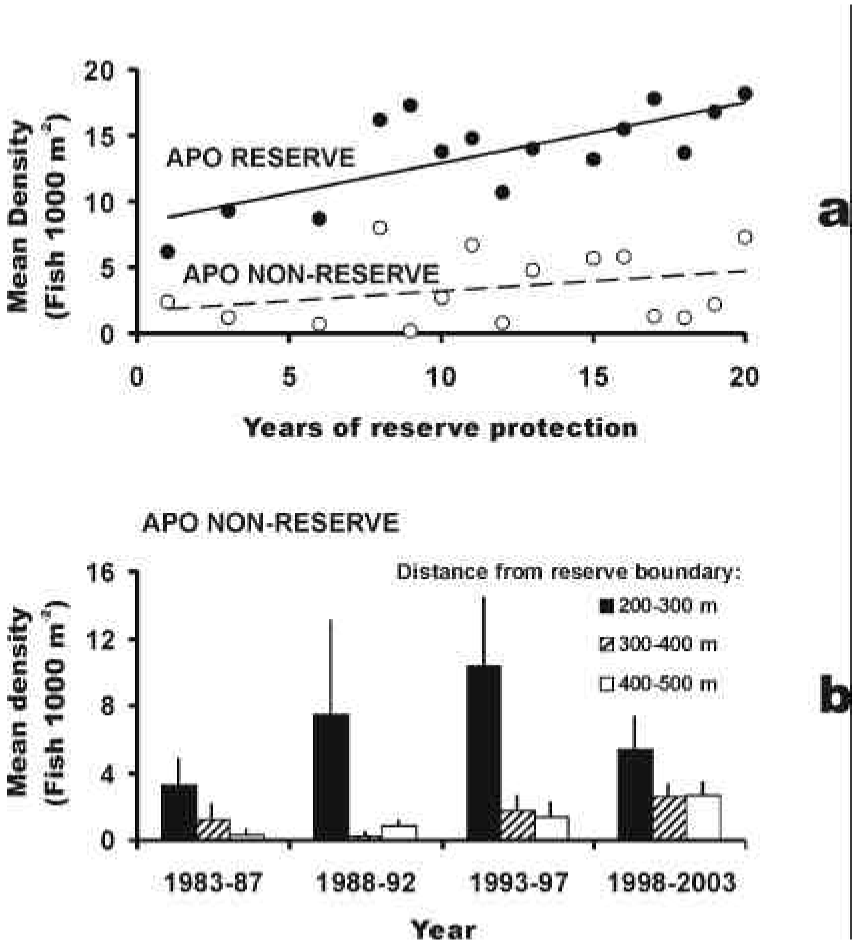


Figure 3.6. (a) Mean density of *Naso vlamingii* in the no-take reserve (filled circles) and a fished non-reserve site (open circles) at Apo Island from 1983 to 2003 (*unpublished data* from G. Russ). Survey depths on the reef slope were reserve: 5-17 m, non-reserve: 9-17 m. The regression line was significant for the reserve (mean density = 0.46 year + 8.31, $R^2 = 0.54$, $F_{1,13} = 15.35$, $p = 0.002$), but not for the non-reserve. (b) Density of *N. vlamingii* at the non-reserve site at different distances from the reserve boundary from 1983-87 to 1998-2003 (*unpublished data* from G. Russ). Error bars are 1 SE. See Figure 3.1 for the position of transects in the non-reserve site surveyed by G. Russ.

Most emigrants probably return to the reserve after movement to fished areas in the short term. The short-term movement from the reserve suggested here was probably temporary, and may be related more to movements of adults to preferred feeding sites within their home ranges. Russ et al. (2003) documented a 3-fold increase in the biomass of *N. vlamingii* on the reef slope inside Apo Reserve over 18 years of protection of the reserve (1983-2001). This increase in biomass was related strongly to an increase in both density and mean size of *N. vlamingii* in the reserve over the period (G. Russ *unpublished data*, Figure 3.6a, 3.7). These findings imply that some individuals of *N. vlamingii* in the reserve could survive to older ages. Since *N. vlamingii* has a long lifespan (about 40 years, Choat & Axe 1996), and local people may have fished for the species close to the reserve (*Tumoy* fishing ground, Figure 3.1) for decades, it is likely that a steady increase in abundance inside the reserve would occur only if some individuals that have come to reside in the reserve have some degree of site-fidelity.

The evidence here suggesting temporary movement of some adult fish from the reserve over the short term is equivocal. Direct observations of individuals showed that movement of *Naso vlamingii* was very limited (<40 m radius). Also, movement of adults (> 26 cm TL) across reserve boundaries was very infrequent. However, these findings were due probably to the short duration of observations (?15 min). The results suggest that movement tended to increase with longer observation time (Figure 3.5b). Nevertheless, direct evidence for the occasional movement of adult fish from a reserve may be obtained from a mark-recapture and/or a tracking study using ultrasonic telemetry. Such studies have been performed at other reserves (Holland et al. 1993, 1996, Zeller & Russ 1998, Meyer et al. 2000, Eristhee & Oxenford 2001). However, such tagging studies may be impractical at Apo Reserve. The reserve was established by the local community for their own benefit. Manipulative research that requires capturing fish in the no-take area may be counterproductive to the efforts of the local community to maintain protection of the reserve (now protected successfully for 20 years) (Russ et al. 2003).

A possible reason for adult fish to emigrate occasionally from a reserve is temporary movement to feeding or mating areas outside the reserve (Roberts & Polunin 1991, Gell & Roberts 2002). In the case of *Naso vlamingii* at Apo Reserve, some adults may prefer to feed at two sites just outside the reserve, about 150-200 m away from each of the lateral boundaries of the reserve. Data from visual census (Figure 3.2) and from experimental fishing (Figure 3.5 a, b) suggest independently that the abundance (density and CPUE) of *N. vlamingii* tends to remain high at these two sites. Data from visual census also suggests that larger adults may have a tendency to migrate towards the southern end of Apo Island. The change in mean density and size, from the first to the second sampling, was greatest at the fished area south of the reserve. Local people consider the southern end of Apo Island, about 150-200 m away from the southern boundary of the reserve, the *Tumoy* fishing ground, as a traditional fishing ground for *N. vlamingii*. Anecdotal information suggests that more *N. vlamingii* tend to be found here

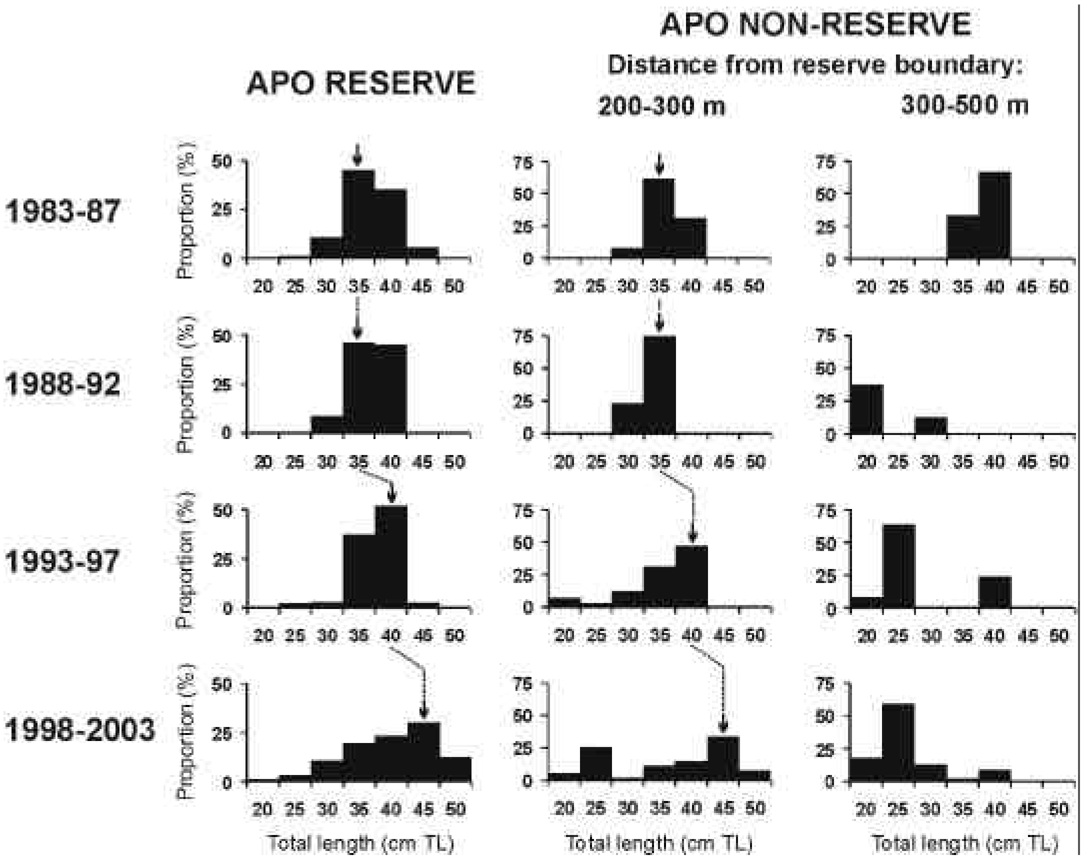


Figure 3.7. Size frequency distribution of *Naso vlamingii* at Apo Island in the no-take reserve and in a fished non-reserve site from 1983-87 to 1998-2003 (*unpublished data* from G. Russ). Survey depths on the reef slope were reserve: 5-17 m, non-reserve: 9-17 m. The size frequency distribution in the non-reserve site is shown for two distance categories from the reserve boundary: 200-300 m (near) and 300-500 m (far). Arrows trace changes in modal size through time. See Figure 3.1 for the position of transects in the non-reserve site surveyed by G. Russ.

than elsewhere, especially when a current from the south is running, or presumably when the fish are feeding (R. Bantaya & G. Mendez, local fishers at Apo Island, *personal communication*). The peaks in abundance at the two sites outside the reserve were probably not due to benthic habitat influences per se (e.g. hard coral cover) (see Chapter 2). However, the pattern of CPUE from experimental fishing may be related strongly to the distribution of planktonic food, since capturing *N. vlamingii* using hook and line depends on when the fish is feeding (when a current is running). Fine-scale circulation patterns that are influenced by local topographic features (Wolanski & Hamner 1988) probably make these two sites just outside Apo Reserve good locations for feeding on plankton.

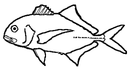
If some larger adults (36-50 cm TL) that are residents in the reserve could emigrate occasionally from the reserve, then why were all of the fish caught just outside the reserve during experimental fishing relatively small (25.6-38.0 cm TL)? One explanation for this seemingly incongruous result is that older (larger) individuals may have learned to avoid baited hooks over time, after escaping from several encounters with them. Anecdotal information from local fishers at Apo Island suggests that larger (older) individuals of *Naso vlamingii* are less interested in the baited hooks than smaller (younger) ones (R. Alaton, R. Bantaya & G. Mendez, *personal communication*). This information is likely to be accurate, since fishers have to watch the fish take the baited hook to know when to pull the line to snag the fish. Local people at Apo Island regard hook and line fishing for *N. vlamingii* as a skill that is difficult to master (M. Pascobello & G. Mendez, Apo Island residents, *personal communication*). During experimental fishing, 8 out of 38 individuals (21%) were able to escape. This figure seems high considering the two fishers who participated in this study are regarded by local people as experienced in this hook and line technique. Nonetheless, all of the individuals that escaped were probably between 0.6-1.0 kg, or 32-40 cm TL only (weights estimated by the two fishers, lengths estimated from the weight-length relationship of the catch during experimental fishing). Evidence from trap fishing also indicates that the smaller sizes of *N. vlamingii* caught using hook and line is an artefact of sampling. Larger *N. vlamingii* (> 40 cm TL) can be caught outside Apo Reserve using large bamboo traps. Local fishers recall an incident in October 2002 where a considerable number of large individuals of *N. vlamingii* up to 1.3 kg (perhaps up to 45-50 cm TL), together with smaller ones < 1.0 kg (weights estimated by local fishers), were caught in two large bamboo traps that were set in *Tumoy* fishing ground (L. Autor, R. Autor, A. Dameles, fishers at Apo Island, *personal communication*).

Other results of this study were more consistent with a longer-term, density-dependent mechanism for spillover, rather than with occasional spillover. A compelling result from experimental fishing was the trend of decreasing mean size of *Naso vlamingii* away from the boundaries of Apo Reserve (Figure 3.3c). Mean size decreased from 34.5 to 32.0 cm TL away from the southern boundary, and from 36.6 to 31.5 cm TL away from the northern boundary. *N. vlamingii* caught by experimental fishing outside the reserve were relatively small adults (25.6-38.0 cm TL). The pattern of decreasing mean size away from the reserve boundaries suggests that these smaller adults may have relocated their home ranges from inside to just outside the reserve over time. Density-dependent spillover is hypothesised to produce a pattern of higher to lower mean size from inside to outside a reserve (Rakitin & Kramer 1996, Kramer & Chapman 1999).

Competition for the best feeding sites or breeding partners among adult *Naso vlamingii* inside the reserve is a probable mechanism that could explain the pattern of decreasing mean size of fish caught as one moves away from the boundaries of Apo Reserve. Furthermore, a size hierarchy may be involved in this competition. Larger adults may eventually drive out the smaller adults from the reserve. The results support this hypothesis. Individuals of *N. vlamingii* seem to become more territorial (display aggressive behaviour and some degree of site-attachment as they grow larger and older (Figures 3.4b, c, 3.5a). Also, more of the larger adults were found inside the reserve (Figure 3.4 a). The proportion of adults displaying aggressive behaviour in the reserve was 40-27%. In the fished area it was just 20%. This suggests that aggressive interactions on a per fish basis were up to twice as frequent in the reserve. Furthermore, when interacting adults differed significantly in size, the larger individual always chased away the smaller one. The sizes of individuals chased by larger fish differed between the reserve and the fished area. In the reserve, fleeing from a larger conspecific was observed only in the 31-35 cm TL size class, and only in the 26-30 cm TL size class outside the reserve. These sizes agree well with the size range of adults caught during experimental fishing just outside the reserve (25.6 to 38.0 cm TL).

The size hierarchy in competitive interactions among adults implies a relationship between potential spillover from the reserve and recruitment of juveniles in the reserve. *Naso vlamingii* probably exhibits an ontogenetic movement from the shallower to the deeper reef slope. Juveniles were found usually on the reef flat or the shallow reef slope, while adults were found usually on the deeper reef slope. This suggests that juveniles which recruit into the reserve will probably move to the same habitat as adults as they grow older. However, individuals that have grown recently into young adults will be smaller than many of the older adults because the latter may survive to larger sizes in the reserve due to protection from fishing. Furthermore, some of the older adults may defend their home ranges in the reserve for several years, even for decades, because of their long lifespan (Choat & Axe 1996). Hence, the smaller (younger) adults that may eventually be driven out of the reserve by larger (older) adults could be members of a cohort of juveniles that recruited to the reserve reef flat several years earlier. Thus, the occurrence and magnitude of spillover events driven by competition inside the reserve may also depend on the frequency and strength of earlier successful recruitment events. Density-dependent spillover may wax and wane in strength in the long-term, driven by recruitment pulses.

In conclusion, the results of this study are consistent with two mechanisms of spillover from a no-take reserve. The first involves the occasional, temporary movement of adult fish from the reserve during daily movements, possibly to preferred feeding sites, within their home ranges. The evidence for this is equivocal since the data were based on visual surveys and only two surveys were made. Few direct approaches (e.g. tagging) were viable in this study to show convincingly that the movement of adult fish across the boundaries of Apo Reserve occurred, because of social factors. The second mechanism involves long-term development of density-dependence. Competition for space, food or mates may result in emigration of smaller adult fish from the reserve. However, the evidence for this, on its own, is also equivocal. This study did not show that competition inside the reserve intensified over time, or that the distribution of smaller adult fish outside the reserve changed over time. However, data collected independently of this study from 1983 to 2003 suggest that the patterns found here, which are consistent with a density-dependent mechanism for spillover, may have developed over time. Long-term monitoring documented a 2-fold increase in the density (a 3-fold increase in biomass) of *N. vlamingii* inside Apo Reserve over 20 years (1983-2003) (G. Russ, *unpublished data*, Figure 3.6a; Russ et al. 2003). Over time, average size of *N. vlamingii* inside the reserve increased from about 35 to 45 cm TL (G. Russ, *unpublished data* 1983-2003, Figure 3.7). This rate of increase in size is consistent with the known growth rate of the species (Choat & Robertson 2003). Outside the reserve, density (and biomass) of *N. vlamingii* increased over time close to the reserve boundary (200-250 m), but not further away (250-500 m) (G. Russ, *unpublished data* 1983-2003, Figure 3.1, 3.6b; Russ et al. 2003). Average size of *N. vlamingii* also increased outside the reserve over time, but only close to the reserve boundary (200-250 m) (G. Russ, *unpublished data*, Figure 3.1, 3.7; Russ et al. 2003). In 2000/2001, hook and line CPUE for *N. vlamingii* was higher closer to the reserve boundaries than elsewhere outside Apo Reserve (Russ et al. 2003). All of this evidence, accumulated over two decades, together with the findings of the present study suggesting net emigration of smaller adult fish from the reserve due to competition, is consistent with the hypothesis of density-dependent relocation of fish from reserve to fished areas.



Chapter 4

How much does the fishery at Apo Island benefit from spillover?

Abstract. Yields caused by net export of adult biomass from reserves, or spillover, are expected to be small. However, few empirical studies have quantified spillover yields, nor responses of fishers to spillover. Fishers may respond to spillover by increasing fishing effort near reserve boundaries. There is some evidence to suggest that catch rates are higher adjacent to reserves because of spillover, but it remains unclear if fishers concentrate effort close to reserves in response to higher catch rates. In this study, the probable spillover contribution of the no-take reserve at Apo Island, Philippines was quantified. The reserve had been well protected for almost 20 years. Spatial patterns of fishing effort, yield, and catch rates of the fishery on the island were documented daily for eight months over two monsoonal seasons. Catch rates and income rates of fishers were found to be highest on the fishing grounds near the reserve. Catch rates near the reserve were higher than in other fishing grounds by a factor of 1.1 to 2.0, depending on the fishing gear. In addition, mean maximum CPUE and IPUE were always higher in the fishing grounds near the reserve. The high catch rates close to the reserve are consistent with the findings of previous studies suggesting that spillover is likely. However, fishing effort was often lowest adjacent to the reserve. Fishing grounds near the reserve accounted for only 3, 15 and 28% of the total hook and line, gill net, and spear gun fishing effort, respectively. Furthermore, fishers seemed to avoid fishing very close to the reserve (within 100 m of reserve boundaries). Fishing effort adjacent to the reserve may have been limited by 1) weather due to monsoons, 2) the traditional importance of fishing grounds far from the reserve, 3) high variability of catch rates and income rates, 4) lower value of targeted species found near the reserve, and 5) social pressures within the local community. The yield taken from fishing grounds close to the reserve comprised only 10% of the overall fishery yield. The actual spillover yield was probably much smaller than this figure. Nevertheless, this study could not detect if the spillover yield of the fishery at Apo Island has yet reached its full potential. Long-term fisheries monitoring is required to determine this.

4.1 Introduction

Fisheries will probably benefit from net export of adult biomass from reserves, or spillover, in a minor way only. Theoretical studies suggest that potential fishery yield (yield per recruit) from spillover would be insignificant, if any at all, except when stock abundance outside reserves is very low due to high fishing mortality (Polacheck 1990, Russ et al. 1992, DeMartini 1993). Even so, potential gains from spillover are predicted to be only moderate. There seems to be general agreement that the more important fishery effect of reserves would be due to net larval export, or the 'recruitment effect' (Carr & Reed 1993, Russ 2002). However, the establishment of reserves, particularly in developing countries, requires strong support from local stakeholders, especially fishers (Russ & Alcala 1996a, Galal et al. 2002). It is a concern that in some cases, support for reserves might be difficult to obtain on the pretext of enhanced fisheries through increased recruitment (Russ & Alcala 1996a). Net larval export may seem less convincing or even ambiguous to fishers because of the broad spatial scale (tens to hundreds of kilometres) at which it will probably occur (Russ & Alcala 1996a, Russ 2002). Yields from spillover, although probably small, may play a critical role in convincing fishers to support establishment and maintenance of reserves (Russ & Alcala 1996a). For some fishers, fish 'spilling-over' from reserves, especially larger fish, will appear to be a more direct and tangible benefit than larvae from distant reserves recruiting to fishing grounds. Thus, spillover may have a bigger positive psychological effect on the attitudes of fishers towards reserves.

Few empirical studies have quantified effects of spillover on fishery yields, nor the responses of fishers to spillover. This is perhaps why some advocates of reserves are sometimes overly optimistic about the potential benefits of spillover. Two studies in one location in Kenya showed that after several years of reserve protection, spillover was not enough to compensate for reduction of total yield due to the creation of a large no-take reserve (McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2000). The reserve, Mombasa Marine Park, took away 50-60% (approximately 6-8 km^2) of the total fishing area (McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2000). In this case the reserve probably occupied too large an area to supplement total fishery yield substantially (McClanahan & Mangi 2000). However, in other countries, no-take reserves as large as Mombasa Marine Park are rare. In the Philippines, for example, reserves that are created with fishery enhancement as a major goal are typically small (<

1 km^2 of reef area) and occupy ? 25% of available local fishing area (Pajaro et al. 1999). To date, only two persuasive studies of Philippine reserves, Sumilon and Apo (approximately 25% and 10% of fishing area, respectively), have shown that spillover yield may affect total fishery yield (Alcala & Russ 1990, Russ et al. 2004). In both cases, however, the magnitude of actual spillover yield was unclear (Russ et al. 1992, DeMartini 1993, Russ et al. 2004). On the other hand, two studies, one in St. Lucia (Roberts et al. 2001) and one in Egypt (Galal et al. 2002), demonstrated that catch rates of fishers increased several years after the creation of networks of reserves. It was argued in these studies that the reserves increased total fishery yield, since catch rates improved but fishing effort remained constant (Roberts et al. 2001, Galal et al. 2002). However, these studies provided no information on total fishery yield, precluding estimation of the magnitude of spillover benefits. More empirical studies are needed to provide assessment of potential spillover effects on adjacent fisheries.

Besides available fishing area, the total fishery yield for a given location will be determined by fishing intensity (Russ 1991). In addition, theoretical studies indicate that spillover yield will be a function of reserve size, fishing mortality rate, and demographic parameters of target species, particularly movement rates (Polacheck 1990, Russ et al. 1992, DeMartini 1993). However, the actual contribution of spillover to total yield will be determined by fishing intensity adjacent to reserves. Indeed it has been suggested that the 'first' sign of spillover developing is fishers fishing close to reserve boundaries (Gell & Roberts 2003). This phenomenon would occur presumably in response to higher catch rates near than far from reserves. Spillover is predicted to produce a pattern of higher abundance of target species outside but close to reserve boundaries, but lower abundance further away (Rakitin & Kramer 1996, Kramer & Chapman 1999, Chapter 2 of this thesis). Decreasing catch rates (an index of abundance) away from reserve boundaries have been demonstrated by experimental trap fishing outside one reserve in Barbados and one reserve in Kenya (Rakitin & Kramer 1996, McClanahan & Mangi 2000). Also, monitoring studies of reserves in Kenya and the Philippines have shown that catch rates of fishers were higher closer to reserve boundaries than further away (McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2000, Russ et al. 2003, 2004). In New Zealand, large catches of lobsters were found to be more common closer to Leigh Marine Reserve than further away (Kelly et al. 2002). However, it is less clear if fishers tend to concentrate more effort adjacent to reserves when spillover may be present. Some studies suggest that fishers may do so

(McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2000). Other studies suggest that fishers do not (Russ et al. 2003, Wilcox & Pomeroy 2003). Some of the evidence for fishers preferentially fishing near reserves is anecdotal (Gell & Roberts 2002). Nevertheless, it is important to stress that intense fishing near the reserve may have the effect of eventually reducing catch rates there (e.g. McClanahan & Mangi 2000).

Fishing effort may not necessarily track the spatial distribution of fish abundance. Fishers may not favour fishing adjacent to reserves even if catch rates there are higher. Catch rates alone may not explain the spatial distribution of fishing effort, because decisions by fishers on where to fish are usually aimed at making a profit. Hence, decisions may be influenced by fishing costs such as fuel or time used up travelling to fishing areas (Hilborn & Walters 1992). These costs will be related to the distance of fishing grounds from home ports or residences of fishers (e.g. Wilcox & Pomeroy 2003). If such costs are negligible, then fishing effort may reflect the spatial pattern of catch rates, provided that fishers have ample information on the latter. However, this situation may be unlikely if fishers keep information about productive areas to themselves (Hilborn & Walters 1992). Furthermore, the strategies of fishers may become highly complex in fisheries that employ a variety of fishing gears and target a multitude of species (Hilborn & Walters 1992). This is likely to be particularly true of coral reef fisheries in developing countries (Russ 1991). Spatial distribution of fishing effort in such fisheries may also depend upon the differential value of target species. In addition, other factors that are unrelated to income may also influence the spatial pattern of fishing effort (Hilborn & Walters 1992). These include weather conditions and social factors, such as local traditions or agreements among stakeholders and managers (Wilcox & Pomeroy 2003). Consideration of the behaviour of fishers in relation to reserves may help determine if reserves are achieving their goal of improving fishery yields through spillover.

The objective of this study was to quantify any spillover contribution of the no-take reserve at Apo Island, Philippines to the total yield of the local fishery. The reserve at Apo Island has been protected for 20 years, and there is evidence to suggest that spillover is present (Russ & Alcala 1996a, Russ et al. 2003, 2004). However, previous studies indicate that access by fishers to productive areas far from the reserve during favourable weather influences the local fishery yield considerably (White & Savina 1987, Bellwood 1988). In the present study, daily fishing effort and yield were documented for eight months covering two monsoonal seasons. Spatial and temporal patterns of fishing effort, yield (biomass and value), and catch rates were compared among fishing grounds around the island and within a few hundred meters of reserve boundaries. A basic assumption of the study is that any spillover supplement of fishes from the reserve at Apo will likely occur within a relatively short distance (300 m) from reserve boundaries. The justification for this is that coral reef fishes in general have relatively small home ranges and limited potential for movement (Zeller & Russ 1998, Chapman & Kramer 2000, Meyer et al. 2000).

4.2 Materials and Methods

4.2.1 Study site

Apo Island (9? 4’ N, 123? 17’ E) is located in the central Philippines, about 7 km southeast of the large island of Negros (Figure 4.1). It is a small volcanic island with a fringing reef area of 1.06 km^2 to the 60 m isobath (0.54 km^2 to the 20 m isobath). The island is inhabited by a community of about 700 permanent residents. The traditional source of income for the community is fisheries. At present, about 100 residents are full-time or part-time fishers. Most fishers use a small, one-person outrigger canoe (called a *banca*) powered by an oar. The major fishing gears used are hook and line, gill nets, spear guns and fish traps. At least 60 species of reef fish, non-reef fish and invertebrates are targeted. The majority of local fishers sell their catch to resident fish buyers at Apo Island. Fish buyers, in turn, bring the catch to Negros to sell in Malatapay town or Dumaguete City.

Apo Island is influenced by a northerly tidal current that is present for most of the year (Figure 4.1). This current presumably carries the food supporting planktivorous fish (e.g. Acanthuridae and Caesionidae) that are abundant on the northern side of the island (Bellwood 1988). Predatory fish such as Carangidae also appear to be common in this area (Bellwood 1988). Local fishers tend to fish the northern side of the island when sea surface conditions are favourable there, mainly targeting the Carangidae. Rough sea surface conditions created by the NE and SW monsoon winds (Figure 4.1) appear to be a significant factor affecting the temporal and spatial pattern of fishing effort at Apo Island (Bellwood 1988). During the SW monsoon (June to September), the south and west sides of the island are dangerous for small boats

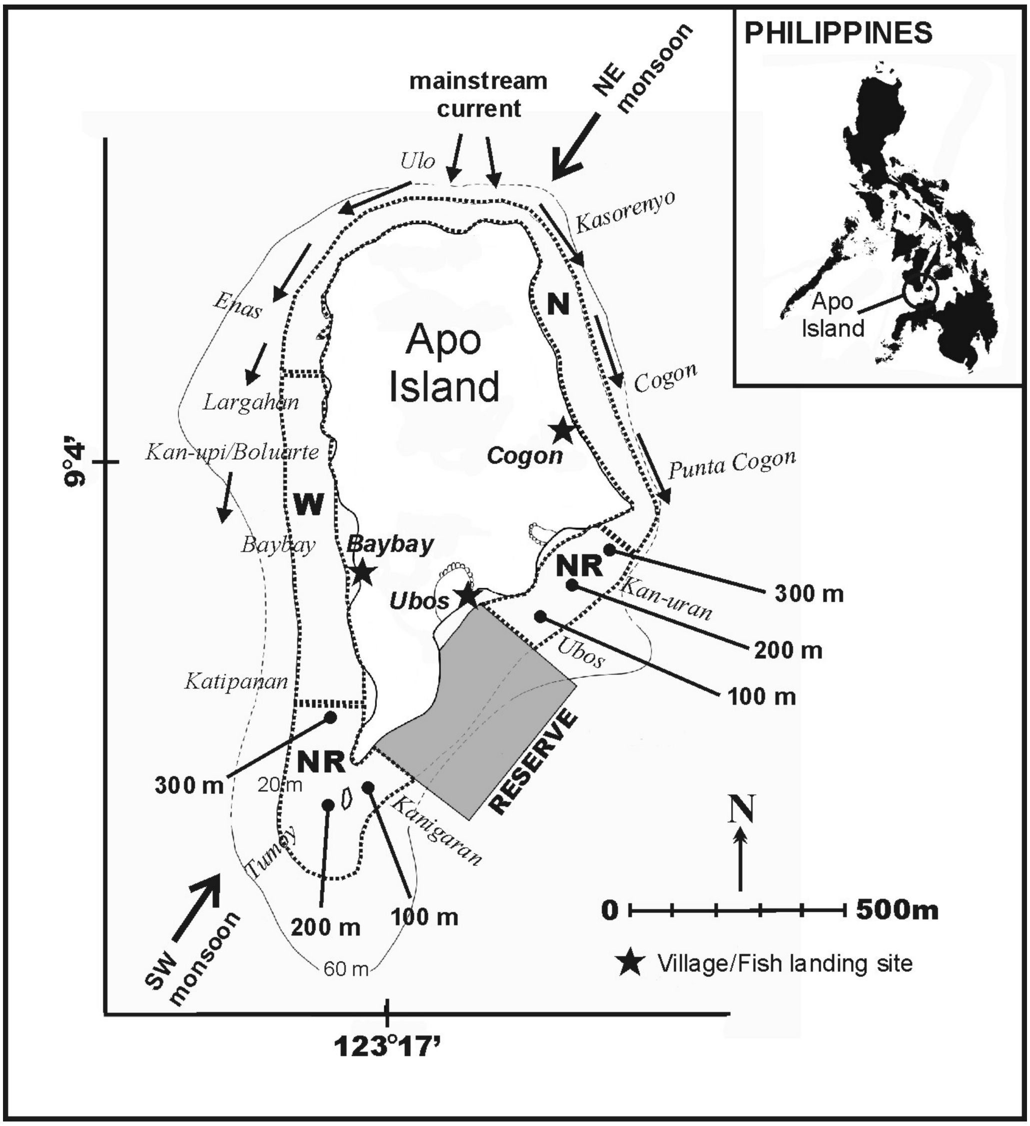


Figure 4.1. Apo Island, central Philippines, showing the no-take reserve (shaded). Names of fishing grounds around the island are italicized. Fishing grounds were grouped into 'northern fishing grounds' (N), 'western fishing grounds' (W), and 'fishing grounds near Apo Reserve' (NR). The reef area of each group of fishing grounds is outlined (0-20 m isobath). Within the fishing grounds near Apo Reserve, the approximate positions of marker buoys used to indicate distance from reserve boundaries are shown. The major fish landing sites are found in *Baybay*, *Ubos* and *Cogon* villages (stars). Large arrows indicate the direction of the monsoons. Smaller arrows indicate direction of the mainstream current.

because of big waves but the northern side of the island is accessible to fishers. During the NE monsoon (November to March), the northern side of the island is treacherous because of big waves, but fishing may be done safely at the south and west sides of the island. Fishing may be done all around the island during the interim calm months (April, May and October).

In 1982, a 450 m long no-take reserve (sanctuary) was established informally by the local community on the southeastern side of Apo Island (Figure 4.1). This reserve occupies approximately 10% of the fringing reef area to the 60 m isobath (about 13% to the 20 m isobath) (Figure 4.1). Establishment of the reserve involved a compromise between avoiding the most important fishing grounds and choosing good coral habitat (Russ & Alcala 1999). The local community has effectively enforced protection of the no-take reserve since 1982 (Russ & Alcala 1999). In 1985/86, the community formally approved a Marine Management Plan that incorporated the no-take reserve (Russ & Alcala 1999, White et al. 2002). The management plan also prohibited destructive fishing methods (e.g. dynamite fishing and muro-ami) and spear fishing with SCUBA. Compliance by local fishers and visitors with these regulations has been generally good. Development of local tourism enterprises has been encouraged by the Marine Management Plan. In the early to mid-1990s, two small resorts that could accommodate recreational SCUBA divers were established on Apo Island. Recently, the local community implemented collection of fees for diving in the no-take reserve and at dive sites around the island.

4.2.2 Data collection

Data were collected from 80 full-time and part-time resident fishers (*Baybay* village – 16, *Ubos* village – 47, *Cogon* village – 17). This group comprised 70-80% of all fishers at Apo Island. The catches of these fishers were recorded daily from July 22, 2003 to February 29, 2004 at the three major fish landing sites on the island (in *Baybay*, *Ubos* and *Cogon* villages, Figure 4.1). Records were kept by three fish buyers, one buyer at each of the landing sites (the houses of fish buyers). The fish buyers recorded the local names and weights of each species or family that comprised the catch sold by each fisher. Catches were weighed on market scales accurate to 0.1 kg. The value of the catch sold by fishers was estimated by multiplying the weight of each species by its average market price per kg in 2003, in Philippine Pesos (PHP). In addition, a resident research assistant (A. Candido) and one of the fish buyers (M. Aldeon) collected daily information from fishers at each of the three villages through interviews. The following information was gathered from fishers: fishing ground/s visited, name/s and number of fishers, date and times when fishing started and finished, fishing gear used, and composition and weight/s of catch (verified from records of fish buyers). Interviewers were able to determine precisely where fishing was conducted because the names and locations of fishing grounds around Apo Island are common knowledge to local people (Figure 4.1). However, if fishing was done near Apo Reserve (e.g. at *Katipanan*, *Tumoy*, *Kanigaran*, *Ubos*, and *Kan-uran*, Figure 4.1), fishers were also asked to estimate how far away they were from the boundaries of the reserve. Colour-coded marker buoys, which fishers could locate and identify easily while fishing, indicated approximate distances from reserve boundaries. These buoys were moored permanently at 100 m (blue), 200 m (red), and 300 m (yellow) from each of the two lateral boundaries of Apo Reserve (Figure 4.1). The buoys were installed with the aid of a GPS receiver and fishers were informed about them one week before the study commenced.

The information here probably reflects accurately the actual patterns of fishing effort and yield of the local market-oriented fishery. The majority of fishers at Apo Island sell their catch to the fish buyers at each of the three fish landing sites monitored (Maypa et al. 2002). Each fish buyer had a group of 'loyal' fishers. Also, most fishers live close to the houses of fish buyers (landing sites). Thus interviewers were able to collect data from fishers daily. Fishing effort and yield data were not adjusted to account for all resident fishers (i.e. any fishers not included in interviews).

Data were not obtained from the following: 1) catch sold to part-time fish buyers on the island, 2) catch sold directly to the main island of Negros, 3) fishers visiting from Negros, 4) catch sold dried, and 5) catch brought directly to homes for consumption. Items 1-4 are probably minor contributors to the total marketed yield. Maypa et al. (2002) surveyed the same fish landing sites monitored in the present study and estimated that only 10% of the total marketed yield (presumably from items 1-4) did not pass through the three major fish landing sites. On the other hand, yield from subsistence fishing (item 5) may be comparable in quantity to the marketed yield (White & Savina 1987), but lower in monetary value. It is unlikely, however, that data on fishing effort collected in this study would differ much from that made on subsistence fishing. It seemed common that fishers went out to sea to catch fish both to sell and keep for personal consumption (R. Abesamis, *personal observation*).

4.2.3 Data analysis

Targeted species were classified into five groups following Bellwood (1988). These were: reef-associated species (Carangidae and Sphyraenidae), reef planktivores (Acanthuridae, Caesionidae, and Pomacentridae), reef species (mainly Lutjanidae, Lethrinidae, Scaridae, Serranidae, Kyphosidae and octopus), open water species (Belonidae, Elopidae and Scombridae), and off-reef species (Lutjanidae). Fishing grounds were classified into three groups. These were: the northern fishing grounds (*Enas, Ulo, Kasorenyo*, *Cogon*, *Punta Cogon*), the western fishing grounds (*Largahan, Kan-upi/Boluarte*, *Baybay*, *Katipanan*) and the fishing grounds near Apo Reserve (Figure 4.1). Data were included in the third group if fishing was done ? 300 m from the reserve boundaries. These three groups of fishing grounds had roughly similar surface areas to the 20 m isobath (northern fishing grounds, 17.3 ha; western fishing grounds, 14.3 ha; fishing grounds near Apo Reserve, 18.9 ha). It was assumed that most fishing was done within or just outside the reef area enclosed by the 0 and 20 m isobaths (Figure 4.1). In addition, data for the fishing grounds near Apo Reserve were classified into the following categories of distance from reserve boundaries: 0-100 m, 100-200 m, and 200-300 m.

Interviews allowed collection of fishing effort data even when fishers returned from trips without catching anything (i.e. fishers who used hook and line, gill nets and spear guns). On average, about 20% of total fishing trips (or 23% of total fishing effort in person hours) returned with no catches. This figure varied considerably according to fishing gears (hook and line: 48%, gill net: 3%, spear gun: 7%) and months. However, fishing effort data for trips with zero catch were collected only beginning in September. Hence, the recorded fishing effort (in person hours) in July and August was adjusted by adding a correction factor in order to account for fishing trips with no catches. Correction factors were calculated from the equation:

$$\mathbf{E_C = [E_R*(E_Z/E_T)]/[1 - (E_Z/E_T)],} \tag{1}$$

where E_C and E_R are the correction factor and recorded monthly fishing effort, respectively, for July or August. E_Z and E_T are the total fishing effort with zero catch and the total fishing effort, respectively, in September. The correction factor was expressed in person hours. Values for September were used because this month is within the same season as July and August (SW monsoon). Correction factors were calculated per gear (hook and line, gill net and spear gun) and per group of fishing grounds. However, prior to adjustment, the recorded monthly fishing effort for July was multiplied by three, to obtain an estimate for a 30-day period. The yield for July was also multiplied by three to obtain a 30-day yield estimate. CPUE (in kg person^{-1} hour^{-1}) per month was calculated by taking the average CPUE of all individual fishing trips made in a given month. Monthly CPUE was calculated per gear, per group of fishing grounds, and per distance from reserve boundaries (0-100, 100-200, and 200-300 m). July and August CPUE were adjusted to account for fishing trips with zero catch, by adding correction factors calculated using equation (1), expressed in number of fishing trips. Hence, correction factors were the number of fishing trips with zero CPUE. Income per unit effort (IPUE) was used as an indicator of relative economic value amongst fishing grounds. This was calculated in the same manner as CPUE, expressed in PHP person^{-1} hour^{-1}. IPUE for July and August were adjusted in the same manner as CPUE.

ANOVA was used to determine: 1) how fishing effort for each of three principal gears [hook and line, gill net and spear gun (fish traps were used rarely)] varied according to seasons (SW monsoon/interim period vs. NE monsoon) and fishing grounds (northern, western, and near Apo Reserve); 2) how CPUE or IPUE varied according to fishing grounds and the three principal fishing gears; and 3) how CPUE varied according to distance from reserve boundaries (0-100, 100-200, 200-300, and > 300 m) and the three principal fishing gears. Monthly estimates of each variate of interest were used as replicates in each ANOVA. Variates were transformed [log (x + 1) or square root (x + 1)] to satisfy ANOVA assumptions. Tukey's test (Zar 1999) was used in all post hoc analyses.

Two sets of indicators of relative economic value were used besides IPUE. The first was the frequency of capturing high-value species, and the frequency of landing a high yield of such species, expressed in number of fishing trips. This was summarised per species group and per fishing ground. High-value species were those with the highest, or the first and second highest, price per kg within species groups. Determination of a 'high yield' within high-value species was more arbitrary. This depended upon the average sizes of individuals within species groups, and whether species were usually landed as individuals or as groups. The following were considered high yield for high-value species: reef associated species (Carangidae, 3 spp.), ? 7.0 kg;

reef planktivores (Caesionidae, 2 spp.), ? 3.0 kg; reef species (Serranidae, 3 spp., Lutjanidae, 7 spp., Lethrinidae, 2 spp.), ? 2.0 kg; open water species (Scombridae, 2 spp.), ? 7.0 kg; off-reef species (Lutjanidae, 1 sp.), ? 2.0 kg. The second was the probability of capturing high-value species, and the probability of landing a high yield of such species, calculated on a per fishing trip basis. This was calculated by dividing the frequencies (the first set of indicators) by the total number of fishing trips that used appropriate fishing gear/s to capture high-value species. Probabilities were calculated per species group and per fishing ground. Yield from traps were excluded in this analysis.

4.3 Results

4.3.1 Seasonal patterns of fishing effort and catch composition among fishing grounds

Hook and line. The majority of hook and line effort (73-98%) was made on the northern fishing grounds from July to December (Figure 4.2a). However, during this period, hook and line fishing on the northern fishing grounds declined steadily (from 2302 to 50 person hours month^{-1}). It remained at low levels from January to February (35-173 person hours month^{-1}) (Figure 4.2a). Hook and line effort on the northern fishing grounds averaged 1015 ? 342 (SE) person hours month^{-1}. The total hook and line yield from this area was 3549 kg (Table 4.1), dominated by reef-associated species (57%, mainly Carangidae) and reef planktivores (24%, mainly *Naso* spp.) (Figure 4.2a). Hook and line effort on the western fishing grounds was much lower, averaging 56 ? 22 person hours month^{-1} (Figure 4.2a). However, in January and February, hook and line effort on the western fishing grounds increased slightly (52-93 person hours month^{-1}). During this period, 38-62% of the total hook and line effort was made on the western fishing grounds, targeting off-reef species [*Aphareus furca* (Lutjanidae)]. The total hook and line yield from the western fishing grounds was only 202 kg (Table 4.1), dominated by off-reef species (37%) (Figure 4.2a). Hook and line effort on the fishing grounds near Apo Reserve was the lowest among fishing grounds, averaging 33 ? 9 person hours month^{-1} (Figure 4.2a). It did not exhibit distinct seasonal patterns. The total hook and line yield near the reserve was only 166 kg (Table 1), dominated by reef species (38%, mainly octopus) (Figure 4.2a).

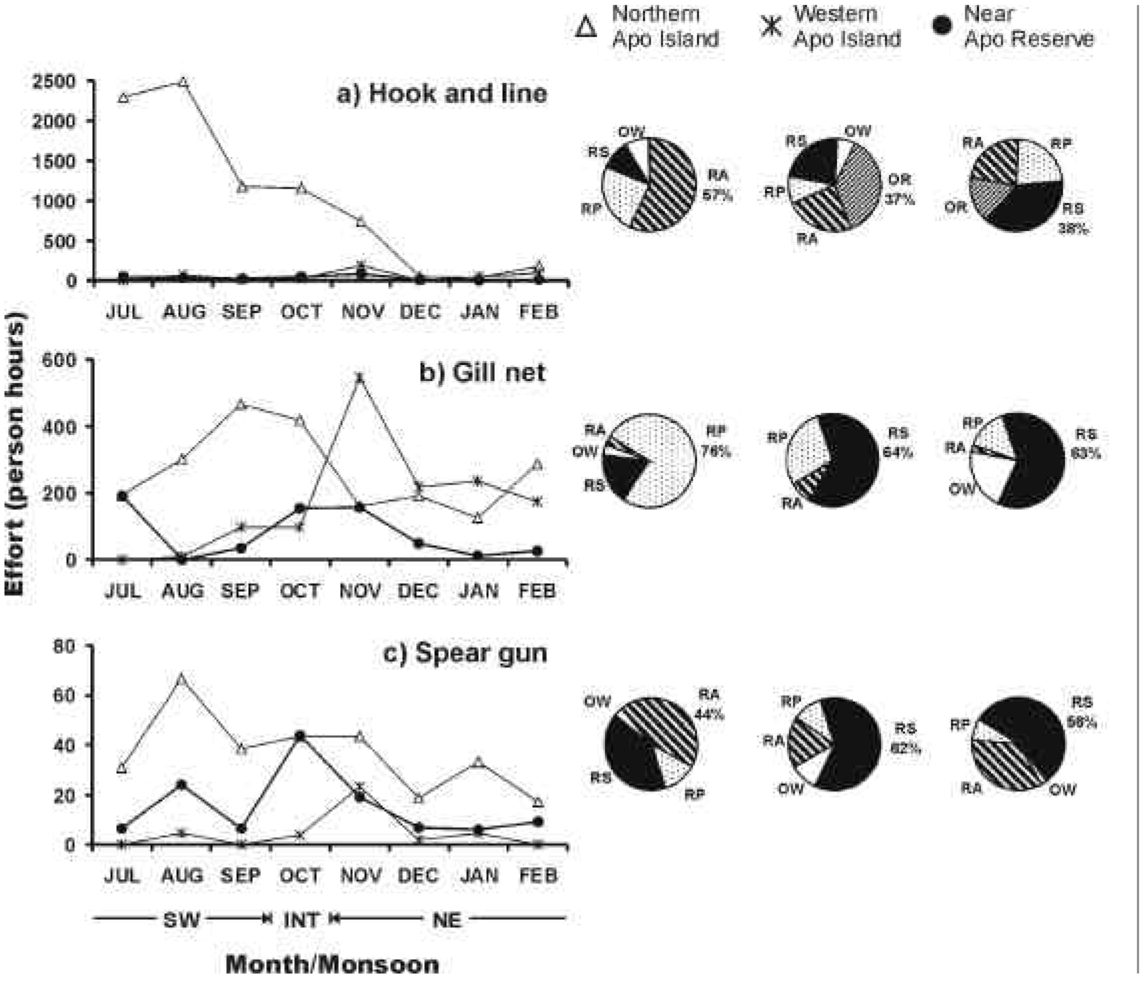


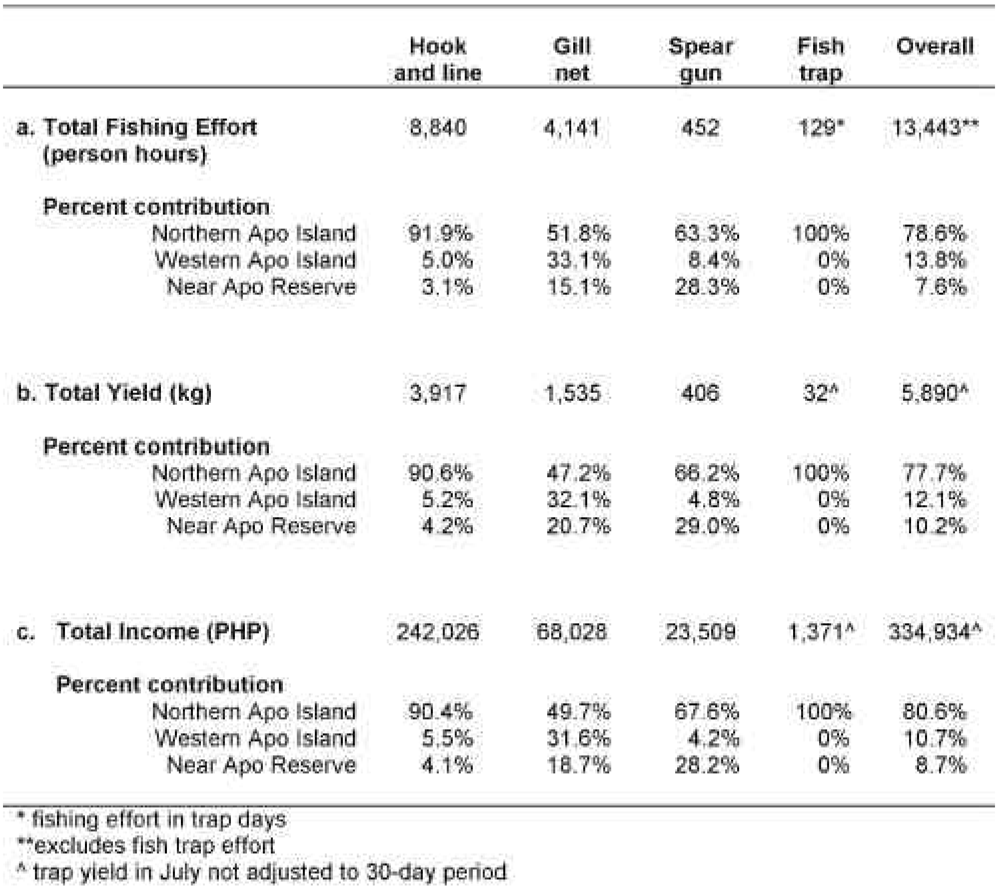
Figure 4.2. Seasonal trends in fishing effort (left hand side) and composition of yield (right hand side) for a) hook and line, b) gill net, and c) spear gun, at the three groups of fishing grounds at Apo Island. Legend is at the upper right hand corner. Seasons: SW - southwest monsoon, INT - calm interim period, NE - northeast monsoon. Species groups: RA - reef-associated species, RP - reef planktivores, RS - reef species, OW - open water species, OR - off-reef species. Percentages of dominant species groups are indicated.

Hook and line effort [square root (x + 1) transformed] differed significantly between seasons ($F_{1,18} = 13.14$, $p = 0.002$) and among fishing grounds ($F_{2,18} = 35.08$, $p < 0.001$). The season x fishing ground interaction was significant ($F_{2,18} = 16.34$, $p < 0.001$). During the SW monsoon/interim period (July to October), hook and line effort on the northern fishing grounds was significantly higher than on the western fishing grounds (Tukey's test: $q_{3,18} = 12.45$, $p < 0.001$) and the fishing grounds near Apo Reserve ($q_{3,18} = 11.84$, $p < 0.001$). During the NE monsoon (November to February), hook and line effort did not differ significantly among fishing grounds. Hook and line effort on the northern fishing grounds was significantly higher during the SW monsoon/interim period (July to October) than during the NE monsoon (November to February) ($q_{2,18} = 9.48$, $p < 0.001$). No significant differences in hook and line effort between seasons were found in the western fishing grounds and in the fishing grounds near Apo Reserve.

Gill net. Gill net fishing occurred mostly on the northern fishing grounds (range: 194-466 person hours month^{-1}) from July to October (Figure 4.2b). Gill net effort on the northern fishing grounds averaged 268 ? 44 person hours month^{-1}. The total gill net yield from this area was 724 kg (Table 4.1), dominated by reef planktivores (76%, mainly Caesionidae) (Figure 4.2b). Beginning in November, gill net effort shifted from the northern to the western fishing grounds. Gill net effort on the western fishing grounds increased dramatically from zero in July, to 545 person hours month^{-1} in November (Figure 4.2b). By November to February, the western fishing grounds accounted for 36-63% of the total gill net effort. Gill net effort on the western fishing grounds averaged 172 ? 62 person hours month^{-1}. The total gill net yield from this area was 493 kg (Table 4.1), dominated by reef species (64%, mainly Scaridae) (Figure 4.2b). Gill net effort near Apo Reserve was much lower compared to other fishing grounds, averaging 78 ? 27 person hours month^{-1} only. However, gill net effort was relatively high near the reserve in July, October and November (193, 155, and 157 person hours month^{-1}, respectively) (Figure 4.2b). The total gill net yield near the reserve was 318 kg (Table 4.1), dominated by reef species (63%, mainly Scaridae) (Figure 4.2b).

Gill net effort did not differ significantly with seasons ($F_{1, 18} = 0.18$, $p = 0.68$) but differed significantly among fishing grounds ($F_{2, 18} = 6.72$, $p = 0.007$). The season x fishing ground interaction was significant ($F_{2, 18} = 7.66$, $p = 0.004$). During the SW monsoon/interim period (July to October), gill net effort on the northern fishing grounds was significantly higher than on the western fishing grounds (Tukey's test: $q_{3, 18} = 5.68$, $p < 0.005$) and on the fishing grounds near Apo Reserve ($q_{3, 18} = 4.81$, $p < 0.01$), but did not differ between the latter two fishing grounds. During the NE monsoon (November to February), gill net effort did not differ between the western and northern fishing grounds, but gill net effort on the western fishing grounds was significantly higher than near Apo Reserve ($q_{3, 18} = 4.47$, $p < 0.025$). Gill net effort on the northern fishing grounds was higher during the SW monsoon/interim period than the NE monsoon [$q_{2, 18} = 2.96$, $p = 0.05$ ($q_{0.05, 2, 18} = 2.97$)]. Conversely, gill net effort on the western fishing grounds was significantly higher during the NE monsoon than the SW monsoon/interim

Table 4.1. Summary of recorded fishing effort, yield and income of the fishery at Apo Island from 22 July, 2003 to 29 February, 2004. Contributions of each of the three groups of fishing grounds (see Figure 4.1) are shown. PHP = Philippine Peso.



	Hook and line	Gill net	Spear gun	Fish trap	Overall
a. Total Fishing Effort (person hours)	8,840	4,141	452	129*	13,443**
Percent contribution					
Northern Apo Island	91.9%	51.8%	63.3%	100%	78.6%
Western Apo Island	5.0%	33.1%	8.4%	0%	13.8%
Near Apo Reserve	3.1%	15.1%	28.3%	0%	7.6%
b. Total Yield (kg)	3,917	1,535	406	32^	5,890^
Percent contribution					
Northern Apo Island	90.6%	47.2%	66.2%	100%	77.7%
Western Apo Island	5.2%	32.1%	4.8%	0%	12.1%
Near Apo Reserve	4.2%	20.7%	29.0%	0%	10.2%
c. Total Income (PHP)	242,026	68,028	23,509	1,371^	334,934^
Percent contribution					
Northern Apo Island	90.4%	49.7%	67.6%	100%	80.6%
Western Apo Island	5.5%	31.6%	4.2%	0%	10.7%
Near Apo Reserve	4.1%	18.7%	28.2%	0%	8.7%

* fishing effort in trap days
**excludes fish trap effort
^ trap yield in July not adjusted to 30-day period

period ($q_{2,\ 18} = 4.67$, $p < 0.005$). Gill net effort near Apo Reserve did not differ significantly between seasons.

Spear gun. Spear fishing occurred mainly on the northern fishing grounds (Figure 4.2c). However, spear gun effort in this area declined from August to February (67 to 17 person hours month^{-1}) (Figure 4.2c). Spear gun effort on the northern fishing grounds averaged 36 ? 6 person hours month^{-1}. The total spear gun yield from the northern fishing grounds was 269 kg (Table 4.1), dominated by reef-associated species (44%, Carangidae) (Figure 4.2c). Spear gun effort on the western fishing grounds was much lower, averaging 5 ? 3 person hours month^{-1} only. It was highest in November (23 person hours month^{-1}) (Figure 4.2c). The total spear gun yield from the western fishing grounds was 19 kg only (Table 4.1), dominated by reef species (62%, mainly Scaridae and octopus) (Figure 4.2c). Spear gun effort near the boundaries of Apo Reserve was also rather low, but was higher on average than on the western fishing grounds (mean 15 ? 5 person hours month^{-1}). It peaked in October (44 person hours month^{-1}) (Figure 4.2c). The total recorded spear gun yield near the reserve was 118 kg (Table 4.1), dominated by reef species (56%, mainly Scaridae and octopus) (Figure 4.2c).

Spear gun effort did not differ significantly between seasons ($F_{1, 18} = 2.12$, $p = 0.16$) but differed significantly among fishing grounds ($F_{2, 18} = 14.26$, $p < 0.001$). The season x fishing ground interaction was not significant ($F_{2, 18} = 1.77$, $p = 0.20$). Spear gun effort on the northern fishing grounds was significantly higher than on the western fishing grounds ($q_{3, 18} = 7.41$, $p < 0.001$) and the fishing grounds near Apo Reserve ($q_{3, 18} = 4.96$, $p < 0.01$). Spear gun effort, however, did not differ significantly between the fishing grounds near Apo Reserve and the western fishing grounds of Apo Island.

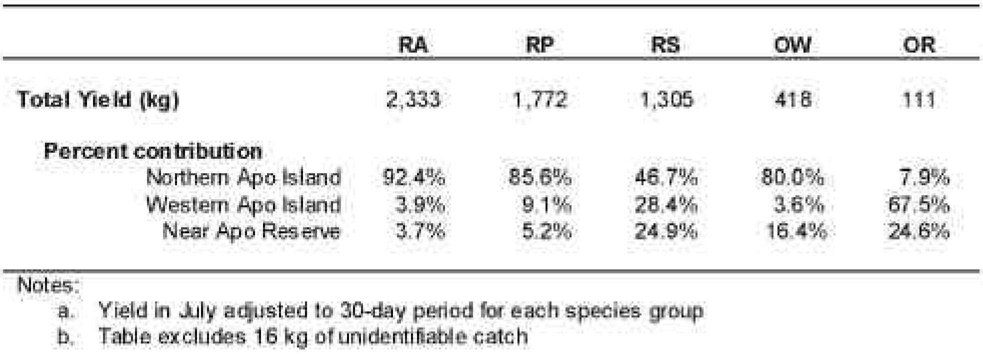
Fish traps. Bamboo fish traps were used on the northern fishing grounds only, and only in July (SW monsoon). The total trap effort was 126 trap days (4 fish traps set for 14 days, 7 for 10 days) with a total yield of 32 kg (Table 4.1). The yield was dominated by reef planktivores (85% Acanthuridae and Caesionidae).

4.3.2 Relative contributions of fishing gears and fishing grounds to overall fishing effort, yield and income

Among the three principal fishing gears, most fishing effort was spent on hook and line, followed by gill nets, then by spear guns (Table 4.1). Hook and line fishing, therefore, contributed the greatest yield and highest income (Table 4.1), accounting for 66% of the total yield and 72% of the total income recorded. Bamboo fish traps contributed the least yield and income (Table 4.1). All types of fishing occurred mainly on the northern fishing grounds (Table 4.1). The northern fishing grounds accounted for 92, 52, 63, and 100% of the total effort for hook and line, gill net, spear gun, and fish traps, respectively. Accordingly, the northern fishing grounds accounted for the vast majority of total yield and total income for all types of fishing gears (Table 4.1).

On the other hand, fishing effort was often lowest on the fishing grounds near Apo Reserve (Table 4.1). The fishing grounds near the reserve accounted for only 3% and 15% of the total effort spent on hook and line and gill net fishing, respectively. These fishing grounds contributed only 4% to the total yield and total income from hook and line fishing, and only 21% to the total yield and 19% to the total income from gill

Table 4.2. Composition of total recorded yield of the fishery at Apo Island from 22 July, 2003 to 29 February, 2004. Contributions of each of the three groups of fishing grounds (see Figure 4.1) are shown. Species groups: RA - reef-associated species, RP - reef planktivores, RS - reef species, OW - open water species, OR - off-reef species.



	RA	RP	RS	OW	OR
Total Yield (kg)	2,333	1,772	1,305	418	111
Percent contribution					
Northern Apo Island	92.4%	85.6%	46.7%	80.0%	7.9%
Western Apo Island	3.9%	9.1%	28.4%	3.6%	67.5%
Near Apo Reserve	3.7%	5.2%	24.9%	16.4%	24.6%

Notes:
a. Yield in July adjusted to 30-day period for each species group
b. Table excludes 16 kg of unidentifiable catch

net fishing. Only 28% of the total spear gun effort was made near Apo Reserve. However, the overall spear gun effort here was about three times higher than that on the western fishing grounds (Table 4.1). The fishing grounds near the reserve contributed about six times the yield and seven times the income of spear fishing on the western fishing grounds (Table 4.1).

Among species groups, the highest yield recorded was for reef-associated species, followed by reef planktivores, then reef species (Table 4.2). Open water and off-reef species were minor contributors to overall yield. Reef-associated species together with reef planktivores accounted for 69% of the overall yield. The northern fishing grounds contributed ? 80% of the total yield of reef-associated species, reef planktivores, and open water species (Table 4.2). Much of the total yield of reef species (47%) was also taken from the northern fishing grounds. The western fishing grounds contributed 68% of the total yield of off-reef species, but accounted for little of the yield for other species group (Table 4.2). The fishing grounds near Apo Reserve often accounted for the smallest contribution to yield for all species groups (Table 4.2). However, about a quarter of the total recorded yield of reef species were taken near the reserve.

4.3.3 Spatial patterns in CPUE and IPUE among fishing grounds

Mean hook and line, gill net, and spear gun CPUE were highest near Apo Reserve (Figure 4.3). CPUE near the reserve was higher than on the northern fishing

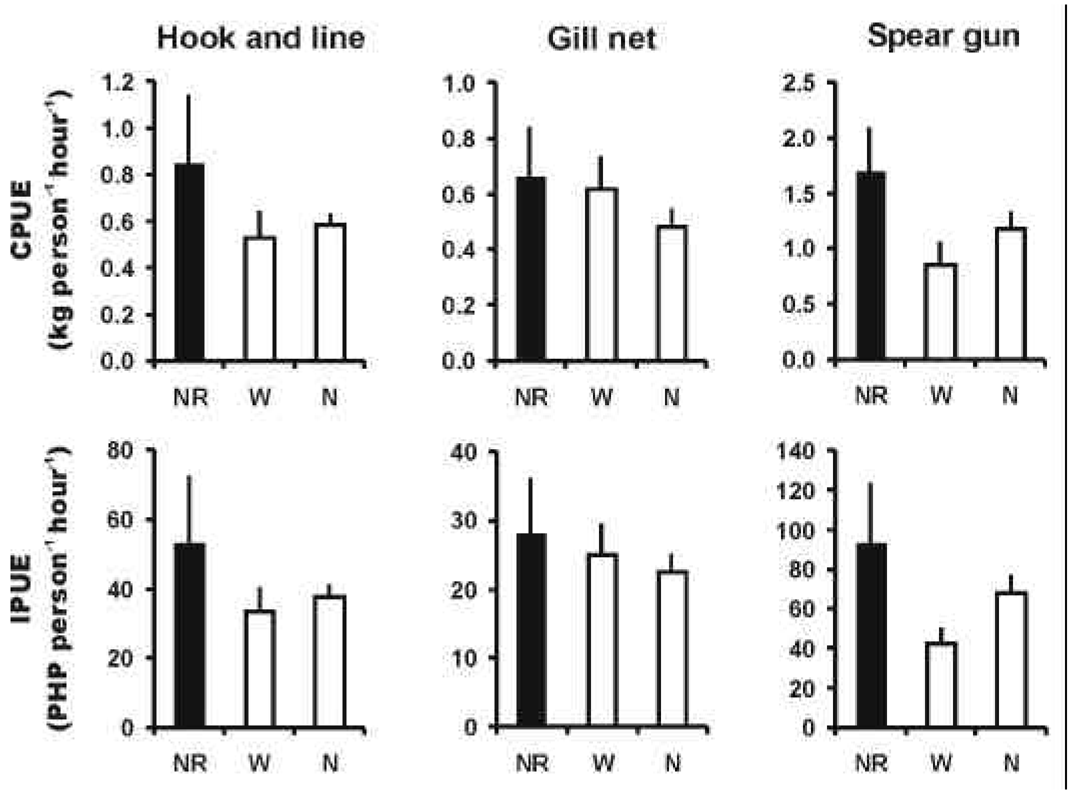


Figure 4.3. Mean catch per unit effort (CPUE) and income per unit effort (IPUE) for each of hook and line, gill net and spear gun at the three groups of fishing grounds at Apo Island. NR - Near Apo Reserve, W - Western Apo Island, N - Northern Apo Island (see Figure 4.1). Error bars are 1 SE.

grounds by a factor of 1.5, 1.4 and 1.4, for hook and line, gill net, and spear fishing, respectively. CPUE near the reserve was higher than on the western fishing grounds by a factor of 1.6, 1.1 and 2.0, for hook and line, gill net, and spear fishing, respectively. However, ANOVA indicated that CPUE [log (x + 1) transformed] did not differ significantly among the three groups of fishing grounds ($F_{2, 57} = 1.87$, $p = 0.16$), but differed significantly among fishing gears ($F_{2, 57} = 9.26$, $p < 0.001$).

The trends in mean IPUE reflected closely that of CPUE (Figure 4.3). Mean IPUE near the reserve was higher than on the northern fishing grounds by a factor of 1.4, 1.2, and 1.4, for hook and line, gill net and spear fishing, respectively. It was higher than on the western fishing grounds by a factor of 1.6, 1.1 and 2.2, for hook and line, gill net, and spear fishing, respectively. However, ANOVA indicated that IPUE [log (x + 1) transformed] did not differ significantly among the three groups of fishing grounds ($F_{2, 57} = 0.49$, $p = 0.62$), but differed significantly among fishing gears ($F_{2, 57} = 6.77$, $p = 0.002$).

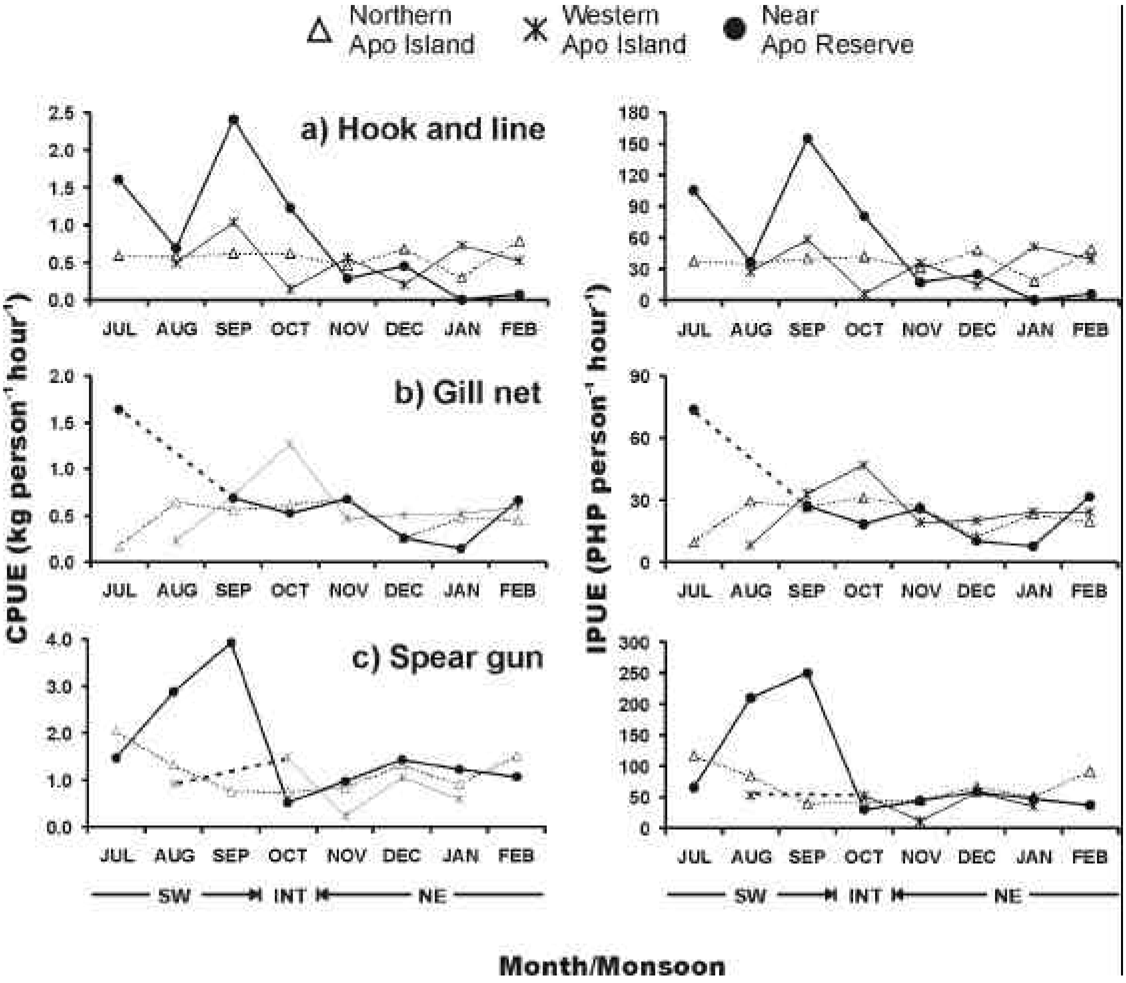
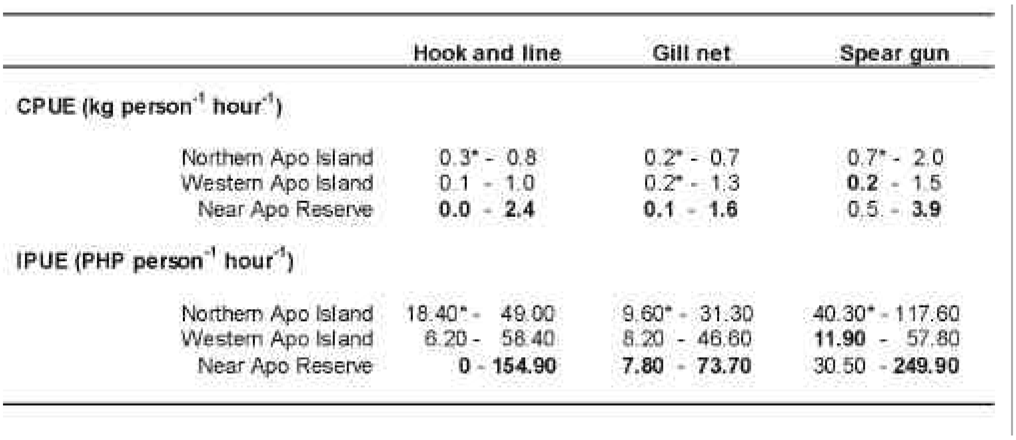


Figure 4.4. Monthly mean catch per unit effort (CPUE) and income per unit effort (IPUE) for a) hook and line, b) gill net and c) spear gun, at the three groups of fishing grounds at Apo Island. Legend is shown at the top of the figure. Seasons: SW - southwest monsoon, INT - calm interim period, NE - northeast monsoon. PHP = Philippine Peso.

Monthly mean hook and line, gill net, and spear gun CPUE and IPUE near Apo Reserve were more variable than in other fishing grounds (Figure 4.4a-c). For example, the monthly hook and line CPUE near Apo Reserve changed from > 1.5, to < 1.0, to > 2.0, and to < 1.5 kg person^{-1} hour^{-1} from July to October. However, from November to February, it remained at < 0.5 kg person^{-1} hour^{-1}, but was zero in January (Figure 4.4a). Monthly hook and line IPUE near Apo Reserve varied accordingly (Figure 4.4a). It changed from > 100, to < 50, to >150, and to < 100 PHP person hour^{-1} from July to October, but was less than 30 PHP person^{-1} hour^{-1} from November to February and was zero in January. In contrast, the monthly hook and line CPUE and IPUE on the northern and western fishing grounds of Apo Island exhibited relatively little variability (Figure 4.4a). For most of the period between July and February, the monthly hook and

Table 4.3. Range of monthly mean catch per unit effort (CPUE) and income per unit effort (IPUE) for each of hook and line, gill net, and spear gun at the three groups of fishing grounds at Apo Island (see Figure 4.1). Lowest minimum and highest maximum average values are in bold letters. Asterisk indicates highest minimum values. PHP = Philippine Peso.



	Hook and line	Gill net	Spear gun
CPUE (kg person^{-1} hour^{-1})			
Northern Apo Island	0.3* - 0.8	0.2* - 0.7	0.7* - 2.0
Western Apo Island	0.1 - 1.0	0.2* - 1.3	**0.2** - 1.5
Near Apo Reserve	**0.0** - **2.4**	**0.1** - **1.6**	0.5 - **3.9**
IPUE (PHP person^{-1} hour^{-1})			
Northern Apo Island	18.40* - 49.00	9.60* - 31.30	40.30* - 117.60
Western Apo Island	6.20 - 58.40	8.20 - 46.60	**11.90** - 57.80
Near Apo Reserve	**0** - **154.90**	**7.80** - **73.70**	30.50 - **249.90**

line CPUE on the northern fishing grounds remained between 0.5 to 0.8 kg person^{-1} hour^{-1}, while IPUE remained between 30 to 50 PHP person^{-1} hour^{-1}. Monthly CPUE for the western fishing grounds remained mostly between 0.2 to 1.0 kg person^{-1} hour^{-1}, while monthly IPUE remained mostly between 15 to 60 PHP person^{-1} hour^{-1}.

Fishing grounds near Apo Reserve often had the lowest minimum monthly CPUE and IPUE for all fishing gears, except for spear gun, which had the lowest minimum monthly CPUE and IPUE on the western fishing grounds (Table 4.3). However, fishing grounds near Apo Reserve always had the highest maximum monthly CPUE and IPUE for all fishing gears (Table 4.3, Figure 4.5). Maximum monthly CPUE near the reserve was higher by a factor of 1.2 to 3.0, depending on the fishing gear and fishing ground (Figure 4.5). Maximum monthly IPUE near the reserve was higher by a factor of 1.6 to 4.3, depending on the fishing gear and fishing ground (Figure 4.5). On the other hand, the highest minimum monthly CPUE and IPUE for all fishing gears were most often found only on the northern fishing grounds of Apo Island (Table 4.3).

4.3.4 Spatial patterns of high-value catches among fishing grounds

Capturing high-value species and landing a high yield of such species were often more frequent on the northern fishing grounds (Table 4.4). For example, 133 captures of high-value reef-associated species (Carangidae – *Caranx ignobilis, C. melampygus,* and *Carangoides* sp.) were made on the northern fishing grounds. In 46 of these

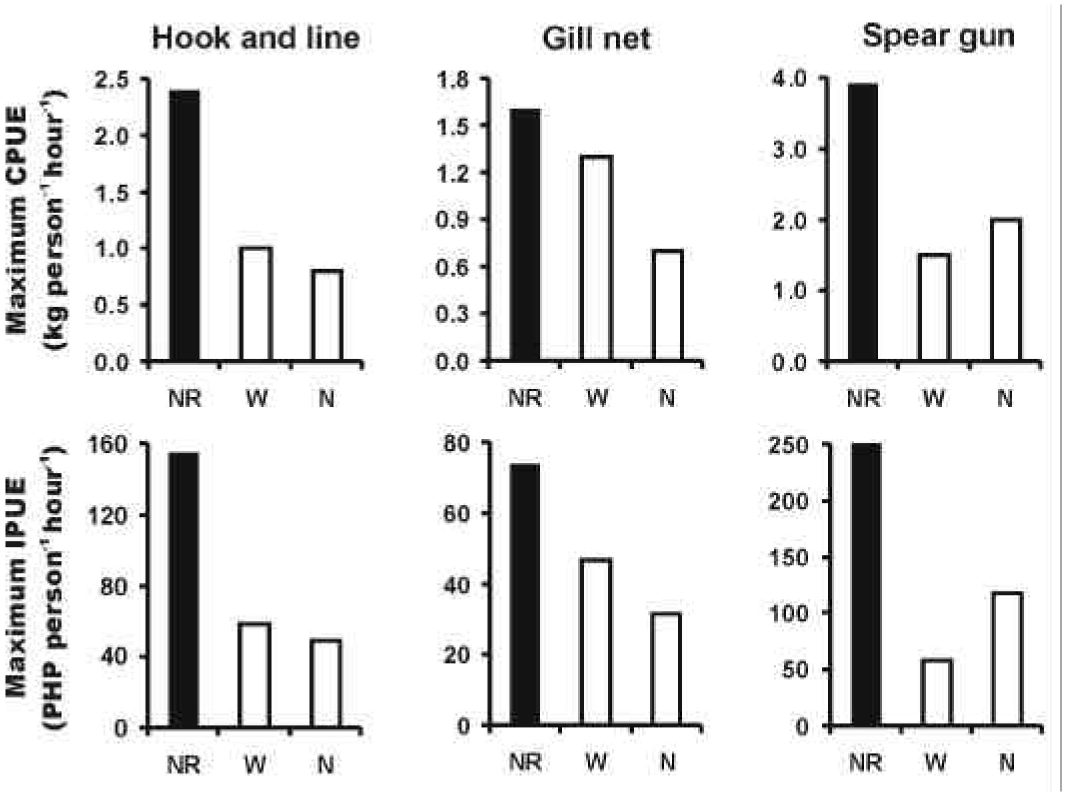
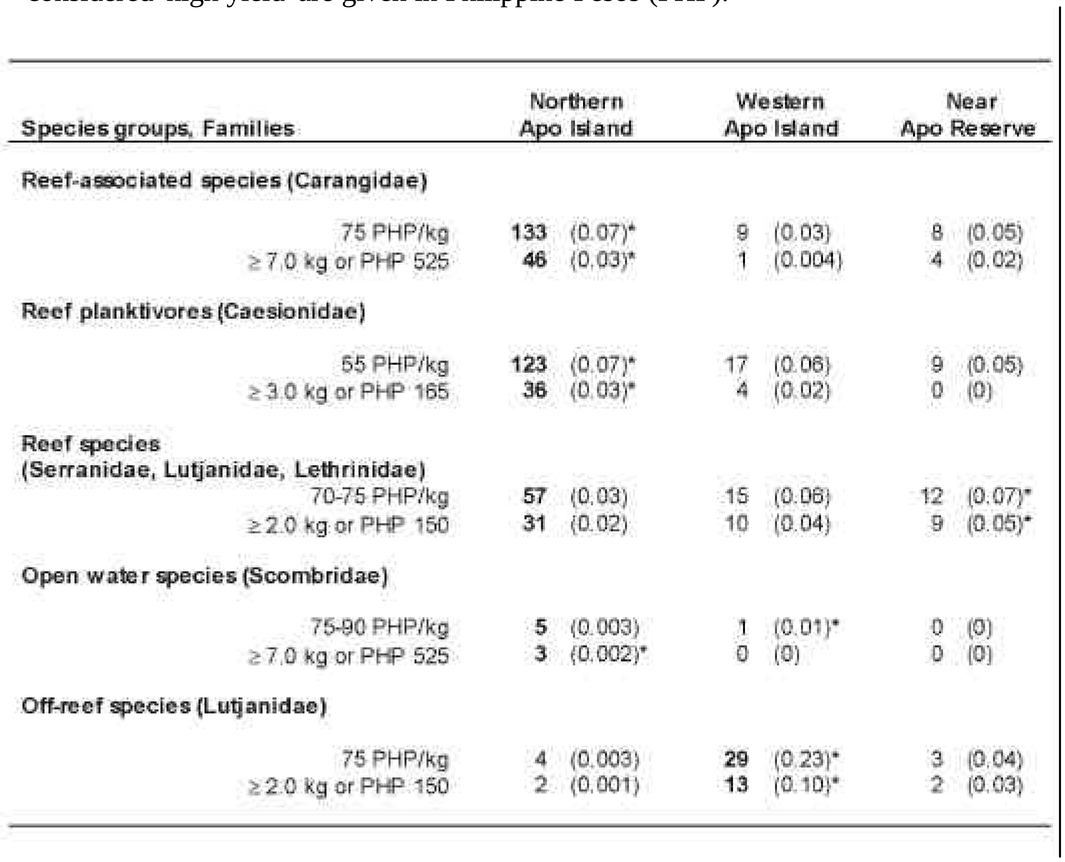


Figure 4.5. Maximum monthly mean catch per unit effort (CPUE) and income per unit effort (IPUE) for each of hook and line, gill net and spear gun at the three groups of fishing grounds at Apo Island. NR - Near Apo Reserve, W - Western Apo Island, N - Northern Apo Island. PHP = Philippine Peso.

captures, the yield was ? 7.0 kg or 525 PHP. Six of the fishes captured weighed ? 15 kg (equivalent to a value of ? 1125 PHP). The largest was a 31.9 kg *C. ignobilis* valued at 2393 PHP, which was also the biggest fish caught during this study. In contrast, only eight captures of high-value reef-associated species were made near Apo Reserve. In only four of these captures were the yields ? 7.0 kg or 525 PHP. Only one of the fishes captured near the reserve weighed > 15 kg. This was a 20.2 kg *C. ignobilis* valued at 1515 PHP.

Capturing high-value species and landing a high yield of such species often had the greatest probability of occurring on the northern fishing grounds (Table 4.4). However, the probability of capturing high-value reef species (Serranidae – *Cephalopholis*, *Epinephelus*, *Variola* spp.; Lutjanidae – *Aprion*, *Lutjanus*, *Macolor*, *Symphorichthys*, *Symphorus* spp.; and Lethrinidae – *Lethrinus* spp.) was greatest near Apo Reserve (Table 4.4). The probability of landing a high yield of such species (? 2.0 kg or 150 PHP) was also greatest near the reserve (Table 4.4). This was despite the

Table 4.4. Frequency and probability of capturing or landing a high yield of high-value species within each species group for each of the three groups of fishing grounds at Apo Island (see Figure 4.1). Upper values: frequency (number of fishing trips) and probability (in parentheses) of capturing high-value species. Lower values: frequency (number of fishing trips) and probability (in parentheses) of landing a high yield of high-value species. Highest frequencies are in bold letters. Asterisk indicates highest probabilities. Prices per kg and prices of what were considered 'high yield' are given in Philippine Pesos (PHP).



Species groups, Families	Northern Apo Island	Western Apo Island	Near Apo Reserve
Reef-associated species (Carangidae)			
75 PHP/kg	**133** (0.07)*	9 (0.03)	8 (0.05)
≥ 7.0 kg or PHP 525	**46** (0.03)*	1 (0.004)	4 (0.02)
Reef planktivores (Caesionidae)			
55 PHP/kg	**123** (0.07)*	17 (0.06)	9 (0.05)
≥ 3.0 kg or PHP 165	**36** (0.03)*	4 (0.02)	0 (0)
Reef species (Serranidae, Lutjanidae, Lethrinidae)			
70-75 PHP/kg	**57** (0.03)	15 (0.06)	12 (0.07)*
≥ 2.0 kg or PHP 150	**31** (0.02)	10 (0.04)	9 (0.05)*
Open water species (Scombridae)			
75-90 PHP/kg	**5** (0.003)	1 (0.01)*	0 (0)
≥ 7.0 kg or PHP 525	**3** (0.002)*	0 (0)	0 (0)
Off-reef species (Lutjanidae)			
75 PHP/kg	4 (0.003)	**29** (0.23)*	3 (0.04)
≥ 2.0 kg or PHP 150	2 (0.001)	**13** (0.10)*	2 (0.03)

higher frequency of capturing high-value reef species, or landing a high yield of such species, on the northern fishing grounds than near Apo Reserve (Table 4.4).

4.3.5 Spatial patterns of CPUE near reserve boundaries

Spatial patterns of CPUE away from reserve boundaries differed among hook and line, gill net and spear gun (Figure 4.6). Gill net CPUE exhibited a pattern of decrease from 0-100 to 200-300 m from reserve boundaries (from 0.88 to 0.70 kg person^{-1} hour^{-1}; Figure 4.6b). Gill net CPUE further decreased > 300 m from reserve boundaries (western and northern fishing grounds, Figure 4.6b). In contrast, hook and line and spear gun CPUE was lowest near (0-100 m) reserve boundaries, but much higher at distances of 100-300 m from the boundaries (Figure 4.6a, c). Hook and line

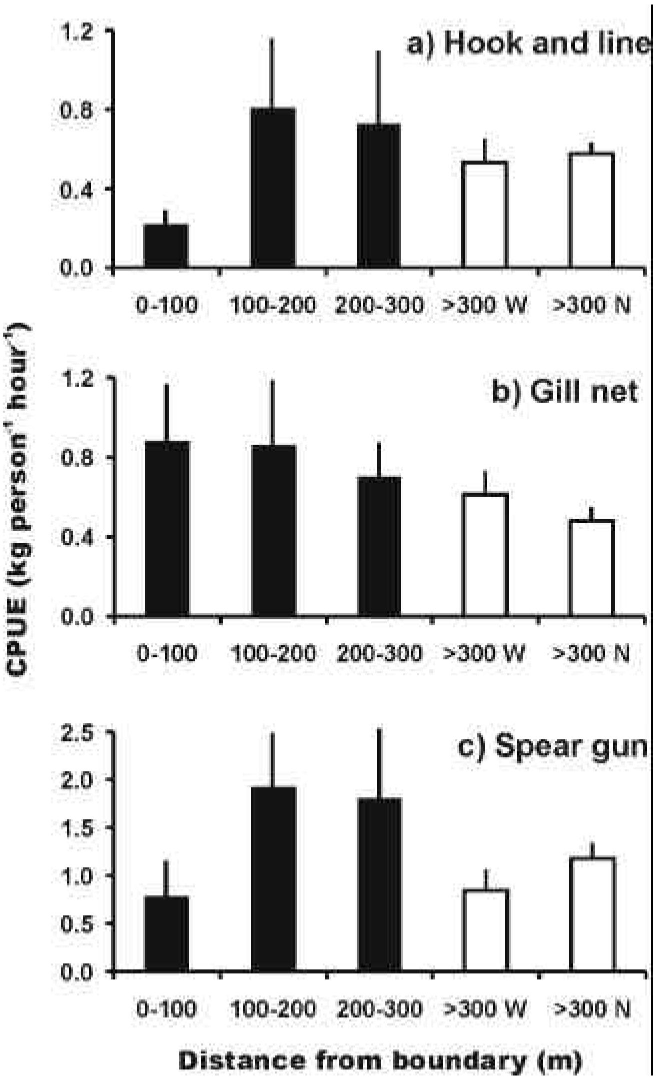
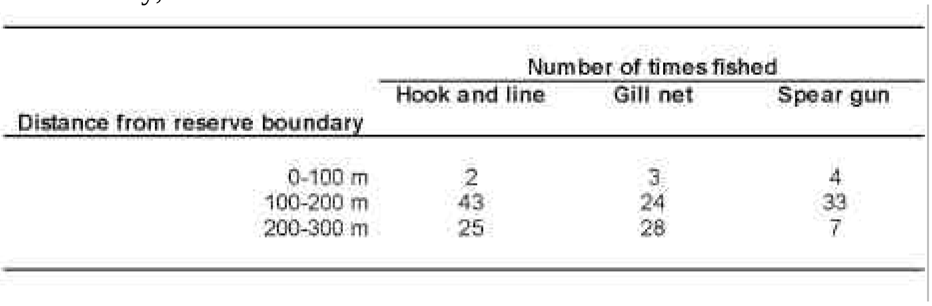


Figure 4.6. Mean catch per unit effort (CPUE) for a) hook and line, b) gill net and c) spear gun at three distances from the boundaries of Apo Reserve [0-100, 100-200, 200-300 m (shaded columns); refer to Figure 1] and at two areas far (> 300 m) from reserve boundaries [W - Western Apo Island fishing ground, N - Northern Apo Island fishing ground (open columns); refer to Figure 4.1]. Error bars are 1 SE.

and spear gun CPUE at 100-300 m from reserve boundaries were higher than further away (> 300 m) from the boundaries (western and northern fishing grounds, Figure 4.6a, c). However, hook and line and spear gun CPUE were highest at 100-200 m from reserve boundaries (Figure 4.6a, c). An ANOVA using data for 0-300 m only indicated that CPUE did not differ significantly with distance from reserve boundaries ($F_{2, 39}$ = 0.73, p = 0.49) or with fishing gears ($F_{2, 39}$ = 2.52, p = 0.09). However, an ANOVA using data for 0-300 and > 300 m indicated that CPUE [log (x + 1) transformed] did not differ significantly with distance from reserve boundaries ($F_{4, 76}$ = 1.19, p = 0.32) but differed significantly among fishing gears ($F_{2, 76}$ = 7.28, p = 0.001).

Contrasting patterns were also found in fishing effort (frequency of fishing trips) at different distances from the boundaries of Apo Reserve (Table 4.5). Hook and line and spear fishing occurred most frequently at intermediate distances from the reserve boundaries, but gill net fishing occurred most frequently at the farthest distances.

Table 4.5. Frequency of fishing trips for each of the three distances from the boundaries of Apo Reserve (see Figure 4.1), for each of hook and line, gill net, and spear gun. Fishing trips were made between 22 July, 2003 and 29 February, 2004.



	Number of times fished		
Distance from reserve boundary	Hook and line	Gill net	Spear gun
0-100 m	2	3	4
100-200 m	43	24	33
200-300 m	25	28	7

However, all types of fishing occurred very infrequently within 100 m of the reserve boundaries (Table 4.5).

4.4 Discussion

Results suggested that the probable spillover contribution of the no-take reserve to the overall yield and income of the fishery at Apo Island was small. Fishing effort was often lowest on the fishing grounds near the reserve. The fishery at Apo Island is primarily hook and line, targeting reef-associated species (Carangidae), and to a lesser extent reef planktivores (*Naso* spp.) (Alcala & Luchavez 1981, White & Savina 1987, Bellwood 1988, Maypa et al. 2002, this study). Ninety-two percent of total hook and line effort was made on the northern fishing grounds of the island. Gill nets, spear guns and fish traps were not used as much as hook and line but fishing with these gears was also done mostly on the northern fishing grounds, contributing to yield of reef-associated species and reef planktivores. Fishing near Apo Reserve contributed mainly to catch of reef species (Scaridae and octopus), which are a less important species group for the local fishery. The fishery depended heavily on access to the northern fishing grounds during the SW monsoon and calm interim period (Bellwood 1988, this study). Results clearly showed that fishing effort on the northern fishing grounds was considerably lower during the NE monsoon, especially for hook and line and gill nets. However, fishing effort did not shift to the fishing grounds near Apo Reserve during the NE monsoon. Gill net fishing seemed to transfer instead to the western fishing grounds, targeting reef species (Scaridae). To a much lesser degree, hook and line fishing also transferred to the western fishing grounds, targeting high-value off-reef species

[*Aphareus furca* (Lutjanidae)]. The northern fishing grounds alone accounted for 78% of overall yield and 81% of overall income recorded in this study (Table 4.1). In contrast, fishing grounds near Apo Reserve accounted for only 10% of overall yield and 9% of overall income (Table 1). However, it is highly unlikely that spillover from Apo Reserve supplied the entire yield taken from fishing grounds near the reserve. That is, the yield near the reserve was probably not composed totally of migrants from the reserve. Therefore, the actual spillover contribution of Apo Reserve would be much less than 10% to the overall yield and income generated by the fishery.

CPUE of fishers was highest on the fishing grounds near Apo Reserve. This pattern may have resulted from 1) spillover from the reserve, or 2) lower fishing intensity near the reserve, resulting in higher abundance of fish and thus higher CPUE. There is more empirical evidence supporting the spillover hypothesis than the lower fishing intensity hypothesis. Considerable evidence exists that some supplement for the fishery near the reserve has developed over time. Firstly, monitoring of Apo Reserve since 1983 has shown that fish populations inside the reserve have increased in abundance, some having tripled in density or biomass over the last two decades (Russ 1985, Russ & Alcala 1989, 1996a, b, 1998a, b, 1999, 2003, Russ et al. 2003, 2004). Secondly, long-term monitoring indicates that the reserve began to export adult fish to a site open to fishing after about 8 years of reserve protection (Russ 1996a, Russ et al. 2003, 2004). Thirdly, catch rates of some species (Acanthuridae) were found to be higher near the reserve than elsewhere around Apo Island after two decades of reserve protection (Russ et al. 2003, 2004). Fourthly, recent studies suggest that patterns of decreasing abundance of some targeted species are present across one boundary of the reserve (Chapter 2 of this thesis). Fifthly, a recent study indicates that density-dependence may be driving net emigration of adult fish (*Naso vlamingii*) from the reserve (Chapter 3 of this thesis). On the other hand, no direct evidence is available to show that fishing effort near Apo Reserve has remained low over the last two decades. However, the fishing grounds near the reserve may be partially exposed to the SW and NE monsoons (Alcala & Luchavez 1981, R. Abesamis *personal observation*). During the SW monsoon, the fishing grounds adjacent to the southern boundary of the reserve may become rough, but those adjacent to the northern boundary are usually calm (Figure 4.1). The pattern is reversed during the NE monsoon. Thus, fishing near the reserve may be reduced for nine months of the year (June to September, November to March) because only one side of the reserve may be fished depending on the monsoon.

The role of the monsoons in limiting fishing intensity near the reserve cannot be ruled out entirely.

If spillover has increased CPUE near the reserve, as long-term evidence suggests, then why have local fishers not responded noticeably to this improvement in catch rates? Weather may be important in limiting fishing effort near the reserve, but it cannot explain the low fishing intensity in this area during the calm interim months. Results show that hook and line and gill net effort on the fishing grounds near the reserve in October were still considerably lower than on the northern fishing grounds (Figure 4.2a, b). Anecdotal information also suggests that most fishers still prefer to fish the northern fishing grounds even during April and May (M. Pascobello, Apo Island resident/Barangay Chairman, *personal communication*). This information is consistent with the findings of previous studies (White & Savina 1987, Bellwood 1988, Maypa et al. 2002), which suggest that the general pattern of fishing effort at Apo Island has not changed much since creation of the reserve in 1982. Furthermore, a reasonable amount of fishing area (8-11 ha) near the reserve is still available to fishers during either monsoon. Given the higher catch rates, fishers could still concentrate effort near the reserve, one side of the reserve at a time, depending on the monsoon. Higher costs [time and energy spent to paddle a *banca*] are certainly not preventing fishers from fishing adjacent to the reserve. The majority of fishers (80%) reside near the reserve, in *Baybay* and *Ubos* villages (Figure 4.1). This also means that for most fishers, the costs of fishing the northern side of Apo Island may actually be higher. It is also unlikely that fishers are not aware of higher catch rates near the reserve. Fishers can probably obtain good information about catch rates from the experiences of fellow fishers, or from word-of-mouth, since the community is relatively small and tightly-knit. Many fishers are members of the same family (including cousins and uncles) or are friends with each other. Income rates per se can also be ruled out as an important influence on the spatial pattern of effort, since results indicated that IPUE for all gears were also highest on the fishing grounds near the reserve. However, one cannot discount the simple reason that the northern side of Apo Island may be regarded by the local community as their main traditional fishing ground. This area has probably been fished for decades, providing the income for families generation after generation.

Other characteristics of the CPUE and IPUE data may also provide a partial explanation. CPUE and IPUE were more variable on the fishing grounds near Apo Reserve (Figure 4). In fact, the fishing grounds near the reserve often had both the highest and lowest CPUE and IPUE per month for all types of fishing gears (Table 4.3, Figure 4.5). The only exception was for spear gun, which had the lowest CPUE and IPUE on the western fishing grounds (Table 4.3). These findings suggest that near the reserve, fishers could obtain very high average monthly catch rates and income rates. However, it also suggests that near the reserve, fishers (using hook and line and gill net) could obtain very low average monthly catch rates and income rates. Monthly CPUE and IPUE were least variable on the northern fishing grounds (Figure 4.4). In addition, the highest minimum average monthly CPUE and IPUE were always found here (Table 4.3). Therefore, a plausible explanation for the spatial pattern of fishing effort at the island scale is that fishers probably tend to avoid fishing near Apo Reserve in order to make their monthly incomes more stable and avoid occasional very low catch rates. Fishers may prefer the northern fishing grounds even if catch rates here are not as high as near the reserve because they are assured of obtaining higher minimum yields and a more stable income. This strategy may be a form of financial 'risk aversion' (Hilborn & Walters 1992). It is further postulated that such a risk aversion strategy could explain why total spear gun effort was higher on the fishing grounds near Apo Reserve than on the western fishing grounds (Tables 4.1, 4.3). Also, a risk aversion strategy may partly explain why fishing with gill nets and hook and line transferred to the western fishing grounds instead of near the reserve during the NE monsoon (Figure 4.2b, Table 4.3).

Other factors may also influence the decision by fishers to concentrate effort on the northern side of Apo Island. For example, the largest and most valuable fish recorded in this study, a 31.9 kg *Caranx ignobilis* worth almost PHP 2400 (about USD 43), was captured from the northern fishing grounds. A high-priced catch such as this one may have a tremendous psychological impact on local fishers. The income from such a yield may be equivalent to about 80% of the sufficient monthly income for one family at Apo Island, which is about PHP 3000 or USD 54 per month (L. Pascobello-Rhodes, Apo Island resident, *personal communication*). In comparison, the largest and most valuable fish caught near Apo Reserve, also an individual of *C. ignobilis*, was only two-thirds the weight (20.2 kg) and value (PHP 1515 or USD 27) of the largest fish caught from the northern fishing grounds. This fish was the only one caught near the reserve with a value greater than PHP 1000 (USD 18). Local fishers probably consider fish of this size more common on the northern fishing grounds. During this study, five individuals of *C. ignobilis* (range: 15.0-19.0 kg) that were captured from the northern fishing grounds had weights similar to the largest fish caught near the reserve.

Therefore, in this case, the biggest psychological impact on fishers may not be due to spillover from the reserve.

Nevertheless, a perception by local fishers that the northern fishing grounds are productive areas for high-value species may not only have a psychological basis. Results indicate that the probability of capturing high-value reef-associated species (Carangidae) and the probability of landing a high yield of such species were highest on the northern fishing grounds (Table 4.4). The same was true for high-value reef planktivores (Caesionidae) (Table 4.4). The probability of landing a high yield of high-value open water species (Scombridae) was also greatest on the northern fishing grounds (Table 4.4). Assuming that fishers prefer to target high-value species, these findings suggest that fishers have better chances of making higher incomes from the northern fishing grounds than from other areas. The better prospect of making a higher income probably influences the decision by fishers to concentrate effort on the northern side of Apo Island whenever it is accessible. Similarly, some fishers probably shift to the western fishing grounds during the NE monsoon (the 'off-season') because of better chances of gaining higher incomes by targeting high-value off-reef species [*Aphareus furca* (Lutjanidae)] (Table 4.4).

The results indicated that the probability of capturing high-value reef species (Serranidae, Lutjanidae, and Lethrinidae), or landing a high yield of such species, was greatest on the fishing grounds near Apo Reserve (Table 4.4). Fishing intensity for high-value reef species, on the other hand, appears to be greatest on the northern fishing grounds, since capturing or landing a high yield of high-value reef species was most frequent here (Table 4.4). Thus, it seemed that most fishers avoided fishing for high-value reef species near Apo Reserve despite better chances of making high incomes from high-value reef species. One plausible explanation for this is that fishers would still prefer to fish the northern fishing grounds since they can target the high-value species that come in larger sizes (Carangidae and Scombridae) or greater numbers (Caesionidae) and at the same time occasionally capture high-value reef-species. This suggests that high-value reef-species (Serranidae, Lutjanidae, Lethrinidae) are only satisfactory alternatives for fishers because they come in smaller sizes (compared to Carangidae and Scombridae) or in smaller numbers (compared to Caesionidae), and therefore would fetch a lower price.

On the other hand, the highest maximum monthly CPUE and IPUE were invariably found near Apo Reserve (Table 4.3, Figure 4.5). This result is consistent with the occasional spillover of bigger fish from the reserve. Such an effect may have a positive influence on the attitudes of fishers towards reserves (Russ & Alcala 1996a). However, in the case of the artisanal fishery at Apo Island, any psychological impact of large catches near the reserve on fishers is probably attenuated by the importance of the northern fishing grounds. In other fisheries, however, occasional spillover of large adults may be important. Recreational 'trophy' fisheries, for example, may benefit directly from such an effect of no-take reserves (Bohnsack 1998, Johnson et al. 1999, Roberts et al. 2001). In New Zealand, Kelly et al. (2002) showed that lobster catch rates (kg trap haul^{-1}) were similar close to and far from Leigh Marine Reserve, even if catches around the reserve consisted of fewer individuals. The lobsters caught near the reserve were bigger. Furthermore, the amount of money made per trap haul close to the reserve was similar to sites far from the reserve.

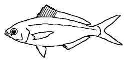
The findings at the scale of a few hundred meters from reserve boundaries provide little evidence to suggest that spillover from Apo Reserve is present. In fact, the most informative result at this spatial scale was that fishers seemed to avoid fishing very close to the reserve, i.e. within 100 m from reserve boundaries (Table 4.5). Furthermore, catch rates seemed to be lowest closest to the reserve boundaries for hook and line and spear fishing (Figure 4.6a, c). However, gill net fishing seemed to have a pattern of decreasing catch rate away from reserve boundaries (Figure 4.6b). Relative gear selectivity interacting with the spatial distribution of target species may explain the differences in spatial patterns of CPUE among the three gears. Gill net fishing is probably less selective than hook and line and spear fishing.

The pattern of fishing effort found very near the reserve can be interpreted in two ways. Firstly, local fishers may well be aware of the relative distribution of catch rates near Apo Reserve, and they adjust their fishing effort accordingly. This interpretation is supported to some degree by the correspondence between the pattern of CPUE and the pattern of fishing effort for hook and line and spear gun. Both CPUE and fishing effort for these gears were highest at intermediate distances (100-200 m), but lower at the closest (0-100 m) and farthest distances (200-300 m) from the reserve (Figure 4.6a, c; Table 4.5). Fishers may know from experience that hook and line and spear gun catch rates are low nearest the reserve (0-100 m), hence they tend to avoid fishing in this area. Experimental fishing using hook and line, but specifically targeting *Naso vlamingii*, supports the contention that hook and line CPUE is low close to the boundaries of Apo Reserve (Chapter 3 of this thesis). Hook and line CPUE for *N.*

vlamingii was higher at intermediate distances (150-200 m) than at the closest (50-100 m) and farthest (250-300 m) distances from the reserve (Chapter 3 of this thesis). Although movement of fish from Apo Reserve to sites at intermediate distances (100-200 m) from reserve boundaries has never been demonstrated directly, research in the last two decades has shown that fish populations (Acanthuridae, Carangidae, Serranidae, Lutjanidae and Lethrinidae) have increased outside but close to one boundary of the reserve (200-250 m) after about eight years of reserve protection (Russ & Alcala 1996a, Russ et al. 2003, 2004). This suggests that the present spatial distribution of hook and line and spear gun effort at the local scale may reflect the response of a small number of fishers to spillover from the reserve.

Secondly, local fishers may avoid fishing very close to the boundaries of Apo Reserve to prevent being accused of poaching inside the reserve. The inconsistent result of a higher catch rate but lower fishing effort nearest the reserve for gill net fishing supports this idea (Figure 6b, Table 5). Furthermore, there may be tremendous motivation for resident fishers not to be implicated in poaching inside the reserve. The ultimate reason for this probably lies in the relative smallness of the community at Apo Island, wherein almost everyone has known each other for most of their lives. The no-take reserve was established by the community for their own benefit (Russ & Alcala 1999). It has been guarded and maintained by community members, many of them fishers themselves, for most of the two decades of its existence (Russ & Alcala 1999). It has an important role in tourism on the island, from which the local community has benefited considerably in many ways (Alcala 1998, Russ & Alcala 1999). It is the principal factor that has made Apo Island a nationally and internationally recognised model for successful community-based resource management (Alcala 1998, Russ & Alcala 1999). Many residents are probably aware of most, if not all, of these achievements. For these reasons, it is clearly against the best interests of a resident fisher, and his family, to be labelled by fellow community members as a threat to the security of their reserve.

In conclusion, this study has shown that spillover yield from the no-take reserve at Apo Island probably contributes much less than 10% of the overall yield of the local fishery. Fishing effort was often lowest near the reserve, despite higher catch rates there. Fishing effort adjacent to the reserve appears to be limited by 1) weather determined by the monsoons, 2) the traditional importance of the northern fishing grounds, 3) high variability of catch rates and income rates, 4) lower value of target species found near the reserve, 5) and social pressures related to the history of community management of the reserve. However, the present study has no information on how fishing effort, yield and catch rates near the reserve have changed over the past 20 years since reserve establishment. Furthermore, it is not clear if spillover yield of the local fishery has reached its full potential or not. Long-term fisheries monitoring is required to answer these questions.



Chapter 5

General Discussion

The findings of this study indicate that spillover from one no-take marine reserve, Apo Reserve, is likely. The main lines of evidence for this are the following. Firstly, patterns of decreasing abundance (density and biomass) of target fishes, from high inside the reserve to low outside, were detected across one boundary of Apo Reserve by underwater visual census. The rate of decrease in abundance for sedentary fishes along this gradient was two to three times greater than that for vagile fishes. This result is consistent with the prediction that the spatial extent of spillover would differ with fish mobility (Rakitin & Kramer 1996). The spatial patterns of decrease in abundance were localised (within 50-200 m of the boundary). They were probably not due to gradients in habitat. Secondly, a pattern of decreasing mean size away from the two lateral boundaries of Apo Reserve (up to 300 m from the boundaries) was found for one target species, *Naso vlamingii*, by experimental fishing. This pattern may have resulted from density-dependent home range relocation of smaller adult *N. vlamingii* from the reserve to adjacent fished areas over the long term. Competitive interactions between adult *N. vlamingii* were up to twice as frequent, on a per fish basis, inside the reserve than outside it. When interacting adults differed in sizes, the larger adult was always observed to chase away the smaller one. The sizes of adults that were chased away (25-35 cm TL) in competitive interactions agreed well with the sizes of those caught by experimental fishing adjacent to reserve boundaries (26-38 cm TL). Thirdly, repeated visual censuses suggested that adults of *N. vlamingii* could move across the boundaries of Apo Reserve within the short term (days). Some adults could probably emigrate occasionally from the reserve, possibly moving to areas suitable for feeding outside the reserve, 150-200 m from the reserve boundaries. However, some larger adults (41-45 cm TL) probably tended to stay inside the reserve. Direct observations suggest that individuals of *N. vlamingii* become more site-attached as they grow older (larger). Lastly, catch rates of local fishers were higher, on average, on fishing grounds near Apo Reserve (within 300 m of reserve boundaries) than on fishing grounds further from the reserve. The highest maximum average catch rates were always found near the reserve, which is consistent with occasional very large catches of target fishes near the reserve due to spillover.

On the other hand, local fishers seemed to have not responded substantially to any spillover from Apo Reserve. Fishing effort was often lowest near the reserve. Spillover probably contributed less than 10% of the total yield of the local fishery. Local fishers probably avoided fishing near the reserve because catch rates there were highly variable. Both the highest and lowest monthly catch rates were often found near the reserve. Most fishing (approximately 79% of overall fishing effort) occurred far from the reserve, on the northern side of Apo Island, where catch rates were lower but less unpredictable. Local fishers may have preferred to fish the northern fishing grounds, instead of those adjacent to Apo Reserve, in order to make their incomes more stable and to occasional avoid very low catch rates (i.e., a risk aversion strategy). In addition, high-value target species (Carangidae, Caesionidae, Scombridae) seemed more abundant on the northern side of Apo Island. Fishers probably concentrated fishing effort on the northern fishing grounds because of greater chances of making more income by targeting high-value species. Fishing effort near Apo Reserve may have also been limited by weather due to the monsoons, the traditional importance of the northern fishing grounds, and social pressures within the local community.

The main conclusion from this study is that Apo Reserve is an example of a successful community-managed reserve that has probably developed spillover, but has provided very limited direct benefits to the local fishery. However, there is little doubt that Apo Reserve has benefited the local community in ways other than spillover. The reserve has become the foundation for a broader, effective program (Marine Management Plan) of fishery conservation and income generation through tourism for the whole island (Alcala 1998, Russ & Alcala 1999, Russ et al. 2004). Fishing practices that are clearly unsustainable, such as use of dynamite and muro-ami (drive net) fishing, have been stopped (Russ & Alcala 1999). Also, the overall standard of living of the local community has improved greatly because of substantial income from tourism (Vogt 1997, Alcala 1998). These indirect effects of reserve establishment may be as important as the direct effects of reserves in managing fisheries in the developing world.

The present study is subject to a range of potential limitations. This study could not provide information on patterns in abundance (density, biomass, mean size or catch rate) and behaviour (movement and aggressive interactions) of target fishes, nor patterns in fishing effort of local fishers, before the reserve was established. Therefore, it could not show if the decreasing gradients of abundance of target fishes adjacent to the reserve developed over time, as abundance increased inside the reserve. It could not demonstrate if density-dependent effects developed over time inside the reserve and caused spillover of target fishes. It also could not show if the frequency of movements of target fish across the boundaries of the reserve increased over time. Most importantly, it could not show if fishers have responded to any increases in abundance of target fishes near the reserve over time. The findings of this study give consistent, yet equivocal, evidence for spillover.

Nevertheless, monitoring since 1983, starting one year after Apo Reserve was created, has shown that density and biomass of several species of target fish have increased inside the reserve over time (Russ 1985, Russ & Alcala 1989, 1996a, b, 1998a, b, 2003, Russ et al. 2003, 2004). Density and biomass of target fishes have also increased over time outside the reserve, but close to one boundary of the reserve (Russ & Alcala 1996a, Russ et al. 2003, 2004). Furthermore, mean size of at least one target species (*N. vlamingii*), which was found in the present study to exhibit size-dependent competition, has increased over the last 20 years inside the reserve (G. Russ, *unpublished data*). Mean size of this species has also increased outside but close to one boundary of the reserve (G. Russ, *unpublished data*). In addition, catch rates of local fishers were found to be higher in recent years (1997/98, 2000/01) compared to the early years of reserve protection (1980/81, 1985/86) (Maypa et al. 2003, Russ et al. 2004). The information collected over two decades of research, together with the results of the present study, strongly suggest that spillover developed over time at Apo Reserve.

Previous studies on the fishery at Apo Island before, and in the two decades after, the reserve was created (Alcala & Luchavez 1981, White & Savina 1987, Bellwood 1988, Maypa et al. 2002) provide no indication of how local fishers may have responded to any spillover from the reserve. Spatial information on yield (biomass and value), fishing effort, and catch rates are virtually absent in these studies. It is unknown whether the patterns in yield, fishing effort, or catch rates in relation to Apo Reserve that were found in the present study are any different from patterns before the reserve was established, or from patterns during the 20 years of existence of the reserve. In general, studies that have attempted to evaluate the effect of no-take reserves on local fisheries have given little attention to the collection of spatially detailed fisheries data (e.g. Roberts et al. 2001, Galal et al. 2002). Only a few studies provide some degree assessment of the fishery effects of reserves in space (e.g. McClanahan & Kaunda-Arara 1996, McClanahan & Mangi 2002). The present study underscored the importance of detailed spatial fisheries data in investigating the impact of spillover on local fisheries.

Future studies on spillover must strive to do the following. Firstly, the density, biomass and size frequency of target species inside and outside reserves, at different distances from reserve boundaries, must be monitored regularly. This monitoring should begin before reserve establishment. Potential effects of changes in habitat on these variates must be accounted for. Secondly, movement of target species across reserve boundaries must be established by tagging or ultrasonic telemetry. The frequency and direction (reserve to fished areas or vice-versa) of movements should be monitored over time, again starting before reserve establishment. Thirdly, behavioural interactions of target species must be monitored inside and adjacent to reserves. This should be related to the data on density, size structure, and movement, to determine if density-dependent emigration from reserves is likely. Fourthly, detailed spatial information on yield (biomass and value), fishing effort, and catch rates in relation to reserves must be gathered. This should be related to the data on density, biomass, size structure, and movement, to determine if fishers respond to any spillover from reserves. All of this monitoring should begin preferably several years before and continue over appropriate time scales, perhaps even decades after reserve establishment.

To unequivocally demonstrate spillover, or any reserve effect, the relevant data on target species and adjacent fisheries must be collected inside and outside replicate reserve and control (fished) sites, before and after creation of reserves (Russ 2002). That is, studies must use a Before-After Control-Impact Pairs (BACIP) design. A study of reserve effects that has used a detailed BACIP design, like the one described above, has never been completed. Perhaps such ideal experiments will be rare. Furthermore, such a study will face enormous funding and logistical limitations, and also social and political constraints (Gell & Roberts 2003). The question of whether reserves can enhance fisheries will probably be answered by large-scale adaptive management experiments that provide indirect but highly plausible evidence for net biomass export from reserves that supplements local fisheries.

References

Alcala AC (1998) Community-based coastal resource management in the Philippines: a case study. Ocean and Coastal Management 38:179-186

Alcala AC, Luchavez T (1981) Fish yield of the coral reef surrounding Apo Island, Negros Oriental, Central Visayas, Philippines. Proceedings of the 4th International Coral Reef Symposium 1:70-73

Alcala AC, Russ GR (1990) A direct test of the effects of protective management on abundance and yield of tropical marine resources. ICES Journal of Marine Science 47:40-47

Attwood CG, Bennett BA (1995) Modelling the effect of marine reserves on the recreational shore-fishery of the south-western Cape, South Africa. South African Journal of Marine Science 16:227-240

Bellwood DR (1988) Seasonal changes in the size and composition of the fish yield from the reef around Apo Island, central Philippines, with notes on methods of yield estimation. Journal of Fish Biology 32:881-893

Bohnsack JA (1996) Maintenance and recovery of reef fishery productivity. In: Polunin NVC, Roberts CM (eds) Reef Fisheries. Chapman and Hall, London, p 283-313

Bohnsack JA (1998) Application of marine reserves to reef fisheries management. Australian Journal of Ecology 23: 298-304

Carpenter KE, Miclat RI, Albaladejo VD, Corpuz VT (1981) The influence of substratum structure on the local abundance and diversity of Philippine reef fishes. Proceedings of the 4th International Coral Reef Symposium, p 495-502

Carr MH, Reed DC (1993) Conceptual issues relevant to marine harvest refuges: Examples from temperate reef fishes. Canadian Journal of Fisheries and Aquatic Sciences 50:2019-2028

Chapman MR, Kramer DL (1999) Gradients in coral reef fish density and size across the Barbados Marine Reserve boundary: Effects of reserve protection and habitat characteristics. Marine Ecology Progress Series 181:81-96

Chapman MR, Kramer DL (2000) Movements of fishes within and among fringing coral reefs in Barbados. Environmental Biology of Fishes 57:11-24

Choat JH, Axe LM (1996) Growth and longevity in acanthurid fishes; an analysis of otolith increments. Marine Ecology Progress Series 134:15-26

Choat JH, Robertson DR (2002) Age-based studies. In: Sale PF (ed) Coral Reef Fishes, Dynamics and Diversity in a Complex Ecosystem. Academic Press, San Diego, California, p 57-80

Clifton KE (1990) The costs and benefits of territory sharing for the Caribbean coral reef fish, *Scarus iserti.* Behavioral Ecology and Sociobiology 26:139-147

Cocheret de la Moriniere E, Pollux BJA, Nagelkerken I, van der Velde G (2002) Post-settlement life cycle migration patterns and habitat preference of coral reef fish that use seagrass and mangrove habitats as nurseries. Estuarine and Coastal Shelf Science 55:309-321

Connell SD, Kingsford MJ (1998) Spatial, temporal and habitat-related variation in the abundance of large predatory fish at One Tree Reef, Australia. Coral Reefs 17:49-57

Craig P (1996) Intertidal territoriality and time-budget of the surgeonfish, *Acanthurus lineatus*, in American Samoa. Environmental Biology of Fishes 46: 27-36

Dahlgren CP, Eggleston DB (2000) Ecological processes underlying ontogenetic habitat shifts in a coral reef fish. Ecology 81:2227-2240

Dayton PK, Sala E, Tegner MJ, Thrush S (2000) Marine reserves: Parks, baselines, and fishery enhancement. Bulletin of Marine Science. 66:617-634

DeMartini EE (1993) Modeling the potential of fishery reserves for managing Pacific coral reef fishes. Fishery Bulletin 91:414-427

Eristhee N, Oxenford HA (2001) Home range size and use of space by Bermuda chub *Kyphosus sectatrix* (L.) in two marine reserves in the Soufriere Marine Management Area, St Lucia, West Indies. Journal of Fish Biology 59:129-151

Froese R, Pauly D (2003) Fishbase. World Wide Web electronic publication. www.fishbase.org. accessed 1 March 2003

Galal N, Ormond RFG, Hassan O (2002) Effect of a network of no-take reserves in increasing catch per unit effort and stocks of exploited reef fish at Nabq, South Sinai, Egypt. Marine and Freshwater Research 53:199-205

Gell F, Roberts CM (2002) The fishery effects of marine reserves and fishery closures. WWF-US, Washington DC, USA

Gell F, Roberts CM (2003) Benefits beyond boundaries: the fishery effects of marine reserves. Trends in Ecology & Evolution 18:448-455

Hilborn R, Stokes K, Maguire JJ, Smith T, Botsford LW, Mangel M, Orensanz J, Parma A, Rice J, Bell J, Cochrane KL, Garcia S, Hall SJ, Kirkwood GP, Sainsbury K, Stefansson G, Walters C (2004) When can marine reserves improve fisheries management? Ocean and Coastal Management 47:in press

Hilborn R, Walters CJ (1992) Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman and Hall, New York

Hixon MA, Beets J (1989) Shelter characteristics and Caribbean fish assemblages: Experiments with artificial reefs. Bulletin of Marine Science 44:666-680

Holland KN, Lowe CG, Wetherbee BM (1996) Movements and dispersal patterns of blue trevally (*Caranx melampygus*) in a fisheries conservation zone. Fisheries Research 25:279-292

Holland KN, Peterson JD, Lowe CG, Wetherbee BM (1993) Movements, distribution and growth rates of the white goatfish *Mulloides flavolineatus* in a fisheries conservation zone. Bulletin of Marine Science 52:982-992

Jennings S (2001) Patterns and prediction of population recovery in marine reserves. Reviews in Fish Biology and Fisheries 10:209-231

Johannes RE (1998) The case for data-less marine resource management: examples from tropical nearshore finfisheries. Trends in Ecology & Evolution 13:243-246

Johnson DR, Funicelli NA, Bohnsack JA (1999) Effectiveness of an existing estuarine no-take fish sanctuary within the Kennedy Space Center, Florida. North American Journal of Fisheries Management 19: 436-453

Jones RS (1968) Ecological relationships in Hawaiian and Johnston Island Acanthuridae (surgeonfishes). Micronesica 4: 309-361.

Jones GP (1991) Postrecruitment processes in the ecology of coral reef fish populations: a multifactorial perspective. In: Sale PF (ed) The Ecology of Fishes on Coral Reefs. Academic Press, p 294-327

Jones GP, McCormick MI (2002) Numerical and energetic processes in the ecology of coral reef fishes. In: Sale PF (ed) Coral Reef Fishes, Dynamics and Diversity in a Complex Ecosystem. Academic Press, p 221-238

Kelly S, Scott D, MacDiarmid AB (2002) The value of a spillover fishery for spiny lobsters around a marine reserve in Northern New Zealand. Coastal Management 30:153-166

Kramer DL, Chapman MR (1999) Implications of fish home range size and relocation for marine reserve function. Environmental Biology of Fishes 55:65-79

Lewis AR (1997) Recruitment and post-recruit immigration affect the local population size of coral reef fishes. Coral Reefs 16:139-149

Lieske E, Myers R (1997) Reef Fishes of the World. Periplus Editions Ltd., Hong Kong. 400 p.

Luckhurst BE, Luckhurst K (1978) Analysis of the influence of substrate variables on coral reef fish communities. Marine Biology 49:317-323

Maypa AP, Russ GR, Alcala AC, Calumpong HP (2002) Long-term trends in yield and catch of the coral reef fishery at Apo Island, central Philippines. Marine and Freshwater Research 53:207-213

McClanahan TR, Kaunda-Arara B (1996) Fishery recovery in a coral-reef marine park and its effect on the adjacent fishery. Conservation Biology 10:1187-1199

McClanahan TR, Mangi S (2000) Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. Ecological Applications 10:1792-1805

Meyer CG, Holland KN, Wetherbee BM, Lowe CG (2000) Movement patterns, habitat utilization, home range size and site fidelity of whitesaddle goatfish, *Parupeneus porphyreus*, in a marine reserve. Environmental Biology of Fishes 59:235-242

Montgomery WL, Myrberg AA, Fishelson (1989) Feeding ecology of surgonfishes (Acanthuridae) in the northern Red Sea, with particular reference to

Acanthurus nigrofuscus (Forsskal). Journal of Experimental Marine Biology and Ecology 132: 179-207

Morgan IE, Kramer DL (2004) Determinants of social organization in a coral reef fish, the blue tang, *Acanthurus coeruleus*. Environmental Biology of Fishes, in press.

Mumby PJ, Wabnitz CCC (2002) Spatial patterns of aggression, territory size, and harem size in five sympatric Caribbean parrotfish species. Environmental Biology of Fishes 63:265-279

Munro JL (1996) The scope of tropical reef fisheries and their management. In: Polunin NVC, Roberts CM (eds) Reef Fisheries. Chapman and Hall, London, p 1-12

Nagelkerken I, van der Velde G (2002) Do non-estuarine mangroves harbour higher densities of juvenile fish than adjacent shallow-water and coral reef habitats in Curacao (Netherlands Antilles)? Marine Ecology Progress Series 245:191-204

Pajaro M, Olano F, San Juan B (1999) Documentation and review of marine protected areas in the Philippines: a preliminary report. Haribon Foundation for the Conservation of Natural Resources, Quezon City, Philippines

Pauly D, Christensen V, Guenette S, Pitcher TJ, Sumaila UR, Walters C, Watson R, Zeller DC (2002) Towards sustainability in world fisheries. Nature 418:689-695

Plan Development Team (1990) The potential of marine fishery reserves for reef fish management in the U.S. Southern Atlantic. Report No. NMFS-SEFC-261, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Centre, Miami: NOAA Technical Memorandum NMFS-SEFC-261

Polacheck T (1990) Year around closed areas as a management tool. Natural Resource Modeling 4:327-354

Rakitin A, Kramer DL (1996) Effect of a marine reserve on the distribution of coral reef fishes in Barbados. Marine Ecology Progress Series 131:97-113

Roberts CM (1997) Ecological advice for the global fisheries crisis. Trends in Ecology and Evolution 12:35-38

Roberts CM, Bohnsack JA, Gell F, Hawkins JP, Goodridge R (2001) Effects of Marine Reserves on Adjacent Fisheries. Science 294:1920-1923

Roberts CM, Ormond RFG (1987) Habitat complexity and coral reef fish diversity and abundance on Red Sea fringing reefs. Marine Ecology Progress Series 41:1-8

Roberts CM, Polunin NVC (1991) Are marine reserves effective in management of reef fisheries? Reviews in Fish Biology and Fisheries 1:65-91

Roberts CM, Polunin NVC (1993) Marine reserves: Simple solutions to managing complex fisheries? Ambio 22:363-368

Robertson DR (1988) Abundances of surgeonfishes on patch-reefs in Caribbean Panama: due to settlement, or post-settlement events? Marine Biology 97:495-501

Robertson DR (1998) Implications of body size for interspecific interactions and assemblage organization among coral-reef fishes. Australian Journal of Ecology 23:252-257

Robertson DR, Polunin NVC, Leighton K (1979) The behavioural ecology of three Indian Ocean surgeonfish (*Acanthurus lineatus*, *A. leucosternon* and *Zebrasoma scopas*): their strategies, and social and mating systems. Environmental Biology of Fishes 4: 125-170

Robertson DR, Gaines SD (1986) Interference competition structures habitat use in a local assemblage of coral reef surgeonfishes. Ecology 67:1372-1383

Rowley RJ (1994) Marine reserves in fisheries management. Aquatic Conservation: Marine and Freshwater Ecosystems 4:233-254

Russ GR (1985) Effects of protective management on coral reef fishes in the central Philippines. Proceedings of the 5th International Coral Reef Congress, p 219-224

Russ GR (1991) Coral reef fisheries: effects and yields. In: Sale PF (ed) The ecology of fishes on coral reefs. Academic Press, p 601-634

Russ GR (2002) Yet another review of marine reserves as reef fishery management tools. In: Sale PF (ed) Coral Reef Fishes. Dynamics and Diversity in a Complex Ecosystem. Academic Press, p 421-443

Russ GR, Alcala AC (1989) Effects of intense fishing pressure on an assemblage of coral reef fishes. Marine Ecology Progress Series 56:13-27

Russ GR, Alcala AC (1996a) Do marine reserves export adult fish biomass? Evidence from Apo Island, Central Philippines. Marine Ecology Progress Series 132:1-9

Russ GR, Alcala AC (1996b) Marine reserves: Rates and patterns of recovery and decline of large predatory fish. Ecological Applications 6:947-961

Russ GR, Alcala AC (1998a) Natural fishing experiments in marine reserves 1983-1993: community and trophic responses. Coral Reefs 17:383-397

Russ GR, Alcala AC (1998b) Natural fishing experiments in marine reserves 1983-1993: roles of life history and fishing intensity in family responses. Coral Reefs 17:399-416

Russ GR, Alcala AC (1999) Management histories of Sumilon and Apo Marine Reserves, Philippines, and their influence on national marine resource policy. Coral Reefs 18:307-319

Russ GR, Alcala AC (2003) Marine reserves: Rates and patterns of recovery and decline of predatory fish, 1983-2000. Ecological Applications 13:1553-1565

Russ GR, Alcala AC, Cabanban AS (1992) Marine reserves and fisheries management on coral reefs with preliminary modelling of the effects on yield per recruit. Proceedings of the 7th International Coral Reef Symposium, p 978-985

Russ GR, Alcala AC, Maypa AP (2003) Spillover from marine reserves: the case of *Naso vlamingii* at Apo Island, the Philippines. Marine Ecology Progress Series 264:15-20

Russ GR, Alcala AC, Maypa AP, Calumpong HP, White AT (2004) Marine reserve benefits local fisheries. Ecological Applications 14:597-606

Sale PF (1978) Reef fishes and other vertebrates: a comparison of social structures. In: Reese ES, Lighter FJ (eds) Contrasts in Behavior, Adaptations in the Aquatic and Terrestrial Environments. John Wiley & Sons, New York, p 313-346

Sanchez Lizaso JL, Goni R, Renones O, Garcia Charton JA, Galzin R, Bayle JT, Sanchez Jerez P, Perez Ruzafa A, Ramos AA (2000) Density dependence in marine protected populations: a review. Environmental Conservation 27:144-158

Shulman MJ (1985) Coral reef fish assemblages: intra- and interspecific competition for shelter sites. Environmental Biology of Fishes 13:81-92

Sladek-Nowlis J, Roberts CM (1999) Fisheries benefits and optimal design of marine reserves. Fishery Bulletin 3:604-616

Sutherland WJ, Gill JA, Norris K (2002) Density-dependent dispersal in animals: concepts, evidence, mechanisms and consequences. In: Bullock JM, Kenward RE, Hails RS (eds) Dispersal Ecology. Blackwell Publishing, p 135-151

Thresher RE (1983) Environmental correlates of the distribution of planktivorous fishes in the One Tree Reef lagoon. Marine Ecology Progress Series 10:137-145

Vogt H (1997) The economic benefits of tourism in the marine reserve of Apo Island, Philippines. Proceedings of the 8th International Coral Reef Symposium. 2:2101-2104

Walters C (2000) Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficacy of marine protected areas: How large should protected areas be? Bulletin of Marine Science 66:745-757

Walters C, Pauly D, Christensen V (1999) ECOSPACE: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems 2:539-554

Wells S, White AT (1995) Involving the community. In: Gubbay S (ed) Marine Protected Areas: Principles and Techniques for Management. Chapman and Hall, London, p 61-84

White A, Savina GC (1987) Reef fish yield and nonreef catch of Apo Island, Negros, Philippines. Asian Marine Biology 4:67-76

White AT, Courtney CA, Salamanca A (2002) Experience with marine protected area planning and management in the Philippines. Coastal Management 30:1-26

Wilcox C, Pomeroy C (2003) Do commercial fishers aggregate around marine reserves? Evidence from Big Creek Marine Ecological Reserve, Central California. North American Journal of Fisheries Management 23:241-250

Williams DB (1991) Patterns and processes in the distribution of coral reef fishes. In: Sale PF (ed) The Ecology of Fishes on Coral Reefs. Academic Press, San Diego, p 434-474

Willis TJ, Parsons DM, Babcock RC (2001) Evidence for long-term site fidelity of snapper (*Pagrus auratus*) within a marine reserve. New Zealand Journal of Marine and Freshwater Research 35:581-590

Wolanski E, Hamner WM (1988) Topographically controlled fronts in the ocean and their biological influence. Science 241:177-181

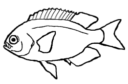
Zar JH (1999) Biostatistical Analysis. 4th Ed., Prentice-Hall, Inc. New Jersey

Zeller DC (1997) Home range and activity patterns of the coral trout *Plectropomus leopardus* (Serranidae). Marine Ecology Progress Series 154:65-77

Zeller DC (1998) Spawning aggregations: patterns of movement of the coral trout *Plectropomus leopardus* (Serranidae) as determined by ultrasonic telemetry. Marine Ecology Progress Series 162:253-263

Zeller DC, Russ GR (1998) Marine reserves: Patterns of adult movement of the coral trout (*Plectropomus leopardus* (Serranidae)). Canadian Journal of Fisheries and Aquatic Sciences 55:917-924

Zeller DC, Stoute SL, Russ GR (2003) Movement of reef fishes across marine reserve boundaries: effects of manipulating a density gradient. Marine Ecology Progress Series 254:269-280



Appendix 1

Publication plan

1. Abesamis RA, Russ GR, Alcala AC (in review) Do gradients of abundance of fish exist across marine reserve boundaries? *Aquatic Conservation: Marine and Freshwater Ecosystems*

2. Abesamis RA, Russ GR (in review) Density-dependent spillover from a marine reserve. Intended journal: *Ecological Applications*

3. Abesamis RA (manuscript in preparation) Movement of adult fish across reserve boundaries over the short-term. Intended journal: *Environmental Biology of Fishes*

4. Abesamis RA, Alcala AC, Russ GR (submitted). How much does the fishery at Apo Island benefit from spillover? Intended journal: *Fishery Bulletin*

5. Russ GR, Abesamis RA, Alcala AC (manuscript in preparation) A review of spillover. Intended journal: *Reviews in Fish Biology and Fisheries*

Appendix 2

Pictures

Plate 1. A large school of *Lutjanus ehrenbergi* encountered in December 2002 inside the no-take reserve in Balicasag Island. A sight such as this one has become rare outside no-take marine reserves in the Philippines (R. Abesamis).

Plate 2. Large predatory reef fishes such as this *Cephalopholis argus* have increased in density and biomass inside the no-take reserve in Apo Island over the last two decades (R. Abesamis).

Plate 3. A large individual of *Naso vlamingii* (Acanthuridae) feeding on zooplankton together with damselfishes inside the reserve at Apo Island. The substantial increase in *N. vlamingii* density inside the reserve over the past 20 years has probably resulted in some spillover to areas very near the reserve. Aggressive interactions between adults may be involved in the spillover of *N. vlamingii* (R. Abesamis).

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Plate 4. The author obtaining length and weight measurements of *Naso vlamingii* caught outside Apo Reserve during experimental fishing using traditional hook-and-line. Experimental fishing was done outside the no-take reserve at increasing distances from the reserve boundaries (A. Pettersen).

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Plate 5. One of the three primary fish buyers at Apo Island (second from right) overseeing the weighing of catch that is to be sold in mainland Negros. Fishing is the traditional source of livelihood at Apo Island. Fishers use hook-and-line, gill nets and spear gun, targeting a multitude of fish and invertebrate species (R. Abesamis).

Plate 6. A diver viewing marine life inside the no-take reserve at Apo Island. Tourism brought by the overall improvement of the marine environment around the island has provided considerable alternative income for the whole community (R. Abesamis).