DOCTOR OF PHILOSOPHY

Stock enhancement of the mud crabs Scylla spp. in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines

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Award date:
2006

Awarding institution:
Bangor University

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A thesis submitted in fulfilment of requirements for the degree of Philosophiae Doctor at the University of Wales, Bangor

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November 2006

ABSTRACT

This study addressed the potential restoration of a population of mud crabs *Scylla* spp. in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines. Survey of the study site showed that 50 of the 70 ha total mangrove area is still natural mangrove and represents suitable habitat for mud crabs. The dominance of *Scylla olivacea* in the mud crab catches, through all life stages, suggests fidelity of this species to the area. The absence of berried crabs in the samples may indicate offshore migration for spawning. The high percentage of immature crabs in all months with peaks in March or April may suggest year-round recruitment with peaks in the summer months. The negatively significant correlation of carapace width (CW) and body weight (BW) to time indicates growth overfishing. Results further revealed low yield and catch per unit effort (CPUE). Population estimates ranging from 14 to 34 crabs ha^{-1} were obtained through a mark-recapture study using coded microwire tag (CWT) and the Jolly-Seber method for open populations. Growth rates were 0.25 mm CW d^{-1} and 1 g BW d^{-1}. Results of the stock enhancement trials showed that small-scale release can increase abundance of mud crabs in a partly isolated mangrove habitat. Growth and survival rates of released crabs suggest that *S. olivacea* is the best suited species for this particular area. The results further revealed that pond-conditioned *S. olivacea* can have higher growth than those released directly from the hatchery and recovery rates equivalent to those of their wild conspecifics. The optimum size-at-release is 65.0-69.9 mm CW regardless of species or source. Both wild and hatchery-reared crabs also exhibited limited post-release movement, supporting the overall conclusion that stock enhancement can be an effective tool in addressing declining fisheries resources.

Lovingly dedicated

to

Nanay,
my guiding star

Tatay,
Inday and her family

and

Pangga

ACKNOWLEDGMENTS

My sincere gratitude …

… to Dr. Lewis Le Vay, my supervisor, for the invaluable help from the very details of planning the experiments to looking for the much needed funds to support my stay in Menai Bridge after the European Commission Project has been terminated; your kindness and friendship is very much appreciated, you simply made everything possible;

… to the European Commission-Culture and Management of *Scylla* spp. (EC-CAMS) Project contract no. ICA4-2000-10225, for funding this degree;

… to Dr. Jurgenne Primavera, my co-supervisor and mentor, for a decade of advice, training, and field exposures; and to the Southeast Asian Fisheries Development Center (SEAFDEC) EC-CAMS Team, Drs. Emilia Quinitio, Fe Estepa, Celia Pitogo, Veronica Alava, and Nerissa Salayo, Edward Rodriguez, Joseph Biñas, Jennette de Pedro, Lillian Gustilo, Quirico Ganon, Joebert Fernando, Ronelio Torrigue, Butch Juanga, Ritchie Andrada and Uldarico Derotas for providing the crabs both from the wild and the hatchery, for the friendship and camaraderie;

… to Dr. Koichi Okuzawa, SEAFDEC Aquaculture Department (AQD) Deputy Chief and Government of Japan-Trust Fund Chair, for funding my trip to the United Kingdom for the *viva voce* examination;

… to Drs. Evelyn Grace de Jesus-Ayson, Jurgenne Primavera and Emilia Quinitio, and Ms. Milagros Castaños, for reviewing the manuscript;

… to Dr. Mark Walton, for helping me out on the use of the microwire tagger, for the references and computer software, for the company and friendship both in Aklan and in Menai Bridge and all the favours big and small;

… to Jon Altamirano and Joseph Biñas, my office buddies, and to Ellen Flor Doyola, Stephen Alayon, Pacita Bantilan, Schedar Novilla, for the hands that never get tired when help was needed;

… to the Aklan group, Josue, Joey, Roey and Roel Inguillo, Simplicio Maylas, Jr., Federico Alonsagay, Renato Crispino, Paquito and Chris Sacapaño, Melecio Pandis, Levi Castor and Jonedel Caparos, the crab collectors in Naisud and Bugtong Bato, Ibajay, for patiently collecting the crabs during sampling period despite the bad weather and schedule during unholy hours; to Fred Segovia, for the untiring tricycle services prioritizing sampling schedule on top of other responsibilities both in the family and in the village, your dedication and support will never be forgotten;

… to the members of the Concerned Scientists and Employees of SEAFDEC AQD (CSESA), for just being there ready to lend a hand and give support; to Drs. Luis Garcia and Wenresti Gallardo for sharing some unpublished information; to Dr. Evelyn Grace de Jesus-Ayson, for the stereoscope which was really a great help;

... to Roel, Rosalie and Roderick Ramos, Wilbert, Wilson, and William Sayson, Allan Retiro, Amzi Albios, Harold Flores, Nino Quimpo, for the assistance during sampling, and tagging of crabs for release, for the good food, the friendship and the camaraderie; special mention to Allan Retiro, for the carved wooden base for the tag holder;

... to Michael Langdown and Kate Hutchinson, for the assistance during the mangrove survey; to John Tapper, for his mud crab data from New Buswang, Kalibo;

... to Mark, Panos, Eva, Marika, Pedro, Camille, Chrissie, Maria, Marija, Natacha, Joanna, Jitka, Julian, Matt, Ibrahim, Ari, Ana, Raquel, Rafa, Dan and Gwyn, for the friendship that made life in Menai Bridge less lonely and grey;

... to Judith Moss and her son William, for the warmth, comfort and kindness during my stay in Menai Bridge and for being my family away from home;

... to Inday and her husband Buds, for the love and support; to the two kids, Vince and Chiqui, for the joy and laughter they bring to our family;

... to Tatay and Pangga, for the love, patience, understanding and support, for the encouragement and prayers that gave me strength to keep going;

... to the Almighty, for making me strong despite the heartaches and headaches that come my way in the early part of this work, for keeping me safe in the midst of century old mangrove forest and in all my travels, for the friendship that developed and flourished, for the people I met who in a way had brighten my day and lighten my load, for the sunshine and the rain, the beaches and the mountains, the falls and the cold spring, the cool breeze and the sound of waves, the frost and the snow, and even the grey, cold and wet days in Menai Bridge that made life more exciting and meaningful...

... I CAN'T THANK YOU ENOUGH!!!

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CHAPTER 1

General introduction

INTRODUCTION

This study aims to evaluate the effects of stock enhancement by release of both wild-caught and hatchery-produced mud crabs, *Scylla* spp., in the partially-degraded mangroves of Naisud and Bugtong Bato, Ibajay Aklan, Philippines. The thesis is divided into five chapters. Chapter 1 reviews the literature relevant to the present study. Chapter 2 describes the topography of the mangrove study area, and includes determination of the area suitable as mud crab habitat and a description of the three creeks used in the study, including levelling in relation to tidal heights. The mangrove flora in each creek are analysed for possible zonation, the diversity between creeks and the density, dominance and frequency of the species between quadrats are compared. Chapter 3 studies the population and fisheries of *Scylla* spp. This includes identifying the most predominant species of *Scylla* in the wild, and determining the sex ratio of the crabs and the size composition of the crabs in the monthly spring catches of the fishers. It compares catches between three gears, their efficiency and selectivity in terms of species, sex and size. It also determines monthly yield and catch per unit effort (CPUE) in terms of number (crab $gear^{-1}$ d^{-1}) and biomass (g $gear^{-1}$ d^{-1}). The information obtained from this chapter serves as baseline information for the stock enhancement trials. Chapter 4 studies the wild population of *Scylla* spp. using a mark-recapture technique employing coded microwire tags and the Jolly-Seber method for analysing open populations. This chapter mainly deals with the abundance, survival and number of new recruits in the population based on the recaptures from the tagged crabs released. From the differences in carapace width (CW) between release and recapture (days), growth rate of wild *Scylla* is also obtained. The conduct of stock enhancement trials, which is discussed in Chapter 5, is based on the findings of the previous chapters. This chapter investigates stock enhancement using both wild-caught and hatchery-reared (unconditioned and conditioned) *Scylla* spp. Even from a small-scale level such as this trial, the effective increase in production following release is reported. Growth rates of the three species of *Scylla* are compared. The need for conditioning of hatchery-reared animals prior to release in the wild to improve survival is identified, and the optimum size-at-release for *Scylla* is determined.

The present chapter reviews literature on mangroves, *Scylla* spp. and stock enhancement. The mangrove section includes topics on mangrove distribution, functions and uses and its associated fauna. The next section is on the mud crabs, *Scylla* spp., being one of the most commercially important mangrove-associated shellfish. This section discusses *Scylla* spp. morphology, taxonomy, distribution, life cycle, habitat, growth and fisheries. The final section covers aspects of stock enhancement including background and rationale, a comparison of the fitness of hatchery-produced vs. wild individuals, release strategies, genetic, diseases and environmental considerations, and assessment and status of stocking initiatives. This chapter ends with the description of the objectives of the succeeding chapters.

MANGROVES

What are Mangroves?

The term mangroves may either refer to the constituent plants of tropical intertidal forest communities or to the whole community itself (Tomlinson 1994). Sometimes, the community of mangrove plants is referred to as 'mangal' and the plant species making up the forest as 'mangroves' (Macnae 1968). Mangroves have been variously described as 'coastal woodlands', 'mangals', 'tidal forests', or 'mangrove forests' (Saenger 2002). They are common on mudflats and banks of tropical and subtropical rivers and coastlines (Spalding et al. 1997), existing at the boundary of two environments (land and water) and receiving nutrients from both terrestrial and aquatic environments. The plants can tolerate salt and brackish water and flourish in heat, salinity and oxygen-starved mud (Macnae 1968). Despite being slow-growing, the ability of mangroves to grow in salt water reduces competition from other plants.

As defined by the possession of morphological, physiological, biochemical and reproductive adaptations that enable them to grow in the unstable and harsh tropic intertidal environment and on their fidelity to the mangrove habitat, approximately 84 species of plants, belonging to 39 genera and 26 families, are recognised throughout

the world as being mangroves (Tomlinson 1994; Field 1995; Duke et al. 1998; Kathiresan and Bingham 2001; Saenger 2002). These mangrove plants can either be trees (*Avicennia* spp., *Rhizophora* spp., *Sonneratia* spp.), shrubs (*Aegiceras* spp., *Lumnitzera* spp.), palm (*Nypa fruticans*) or ferns (*Acrostichum* spp.). Of these 26 families, only two are exclusively mangroves and dominate mangrove communities throughout the world. These are the families Rhizophoraceae and Avicenniaceae (Hogarth 1999). The widest and the most diverse mangrove forests are found in South and Southeast Asia. Excluding ferns, 35 species are found in the Philippines, 33 of which are known in Panay Island (Primavera et al. 2004), and 27 have been documented in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, where the present study was conducted.

Mangroves possess characteristics that make them structurally and functionally unique (Hogarth 1999; Alongi 2002; Saenger 2002). Tomlinson (1994) described the characteristics of the major elements of the mangrove community or the so-called "strict or true mangrove" species. True mangroves possess all or most of these features: 1) complete fidelity to the mangrove environment, 2) major role in the structure of the community and the ability to form pure stands, 3) morphological specialization that adapts them to their environment, 4) some physiological mechanism for salt exclusion to grow in sea water, and 5) taxonomic isolation from terrestrial relatives. True mangroves occur only in the mangrove forest and do not extend into the land. Due to differences in environmental tolerances, they tend to form zonation patterns revealing distinct species associated with the lower and upper intertidal zones, occupying zones where optimum growth can be attained.

Morphological and ecophysiological characteristics and adaptations of mangrove trees include aerial roots, viviparous embryos, tidal dispersal of propagules, rapid rates of canopy production, frequent absence of an understorey, absence of growth rings, wood with narrow, densely distributed vessels, highly efficient nutrient retention mechanisms, and the ability to cope with salt and to maintain water and carbon balance (Popp 1995; Alongi 2002). On dealing with high salt concentrations, they are capable of salt exclusion (Scholander et al. 1962; Popp 1995), salt extrusion (Boon and Allaway 1986; Balsamo and Thomson 1993), salt storage (Wang and Lin 1999),

succulence (Sobrado 2000), compartmentalization (Mizrachi et al. 1980), and osmocompensation (Downton 1982).

The most prominent feature of the mangrove ecosystem is the varying and complex root system of the different species of mangroves that thrive in it. These plants create a unique environment, especially from the intricate designs of their aerial roots associated with gas exchange. According to Saenger (2002), the root system of many mangroves display morphological, physiological and metabolic adaptations which aid in overcoming problems associated with their anaerobic environment. Above ground root types include 1) pneumatophores – pencil-like structures or conical projections arising from the cable root system and extending into the air (*Avicennia* spp., *Sonneratia* spp., *Xylocarpus moluccensis*); 2) knee roots – modified sections of the cable root system which first grow upward above the substrate then downward back to the substrate (*Bruguiera* spp., *Ceriops* spp.); 3) prop or stilt roots – branched roots that arise from the trunk and grow into the substrate (*Rhizophora* spp.); 4) buttress roots – similar to stilt roots in origin but expanding into flattened blade-like or ribbon-like structures (*Heritiera littoralis*, *Xylocarpus granatum*); and 5) aerial roots – unbranched roots arising from the trunk or lower branches and descending downward but usually not reaching the substrate (*Rhizophora* spp., *Avicennia* spp.) (Tomlinson 1994; Hogarth 1999; Saenger 2002). Aside from variations in root structures, adaptations include lenticels or tiny pores for gas exchange found on the surface of the tree trunks or on the roots. These structures are very prominent and in great numbers in *Camptostemon philippinensis*, from the exposed roots to the trunks and up to the branches of these trees.

Mangroves display unique methods of reproduction; viviparity, cryptoviviparity, normal germination on soil and vegetative propagation. Vivipary is the precocious and continuous growth of offspring while still attached to the maternal plant (Thomas and Paul 1996). This reproductive strategy allows the embryo to develop, rupture the pericarp and grow beyond it while still attached to the parent tree. Cryptovivipary, on the other hand, allows the embryo to develop within the fruit but not to the extent of enlarging sufficiently enough to rupture the pericarp (Saenger 2002). According to Smith and Snedaker (1995), this reproductive pattern allows seedlings to develop some tolerance to salinity before being released from the parent plant. Other species

develop by simple germination and do not germinate while still on the parent tree. Despite the differences in their methods of reproduction, whether viviparous or non-viviparous, most species have similar buoyancy, rate of root and shoot initiation and tolerance to salinity (Clarke et al. 2001).

Mangrove Distribution

Mangroves are distributed circumtropically, occurring in 112 countries and territories (Kathiresan and Bingham 2001), covering an estimated area of 181,077 km^2, 41% of which are in South and Southeast Asia (Spalding et al. 1997). The largest single area of mangroves in the world is found in the Bangladesh part of the Sunderbans, which covers an area of almost 600,000 ha including waterways (Bandaranayake 1998). Tomlinson (1994) described the distribution of mangroves in the eastern and the western taxa with no species overlap. The east has approximately three times the number of species as the west. Duke (1992), on the other hand, divided mangrove species into two global hemispheres, the Atlantic East Pacific and the Indo-West Pacific with the latter having four times the number of species found in the former (Fig. 1.1).

Mangrove distribution is determined by latitude, temperature (air temperature >20°C; water temperature ≥24°C) and coastal aridity (Tomlinson 1994). According to Spalding et al. (1997), rainfall has also a strong influence over mangrove distribution, largely by its effect on salinity. Mangroves are almost exclusively tropical (Hogarth 1999) and are confined between 30°N and 30°S latitudes. However, there are outliers to the north in Bermuda, southern Florida and southern Japan and to the south in South Africa, Australia and New Zealand (Tomlinson 1994). The Hawaiian Islands have six species of mangroves, all of which were introduced since the 1900s (Kathiresan and Bingham 2001). On a global scale, the essential environmental prerequisites for mangrove development are temperature, mud substrate, protection, salt water, tidal range, ocean currents, and shallow shores. Mangroves develop best in tropical estuaries which receive heavy rainfall evenly distributed throughout the year, whereas aridity is a limiting factor in many regions of the world (Blasco 1984). In her 19th century travels around tropical West Africa, Mary Kingsley noted that mangroves were able to grow on sand, peat, rock and coral, but the most extensive and luxuriant

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Figure 1.1. Mangrove distribution in six biogeographic regions belonging to two global hemispheres (Atlantic East Pacific and Indo-West Pacific) showing the number of mangrove species found in each region. (Adapted from Duke N.C. 1992. Mangrove floristics and biogeography. In: Robertson A.I., Alongi D.M. (Eds.) Tropical Mangrove Ecosystems. American Geophysical Union, Washington, DC, pp. 63-100).

mangroves were associated with mud and muddy soils (Saenger 2002). Large scale currents may also influence distributions by preventing propagules from reaching some areas because different species differ in the length of time their propagules remain viable, establishment success, growth rates and tolerance limits (De Lange and De Lange 1994). Mangroves are typically distributed from mean sea level to highest spring tide, and perhaps the most conspicuous feature on first glance is the sequential change of tree species parallel to the shore.

Mangrove zonation, described as the sequence of serial communities from seawards to landwards, has been categorized into parallel zonation which is commonly observed along shorelines and longitudinal zonation which is the common upriver zonation (Saenger 2002). Parallel zonation of mangroves along coastlines, often characterized by monospecific zones, has been reported in Asia (Macnae 1968; Satyanarayana et al. 2002), Australia (Smith 1992), the Americas (McKee 1995; Vegas-Vilarrubia 2000), and Africa (Macnae 1968; Matthijs et al. 1999). Longitudinal zonation is the definite sequence of plant communities along the course of a river estuary. Longitudinal upriver zonation has been described in the creeks in Inhaca Island (Macnae and Kalk 1962), along the Shiira River in Iriomote Island (Kuraishi et al. 1985), and in the Daintree River in Queensland and in rivers along the Atlantic and Pacific coasts of Panama (Duke et al. 1998). Macnae (1968) discussed three schemes proposed to describe mangrove zonation which are based on frequency of inundation by Watson (1928), salinity of soil water with tidal flooding as a secondary factor by De Haan (1931), and dominant tree species by Walter and Steiner (1936). Many factors have been suggested to account for the apparent zonation of trees and other associated organisms across the intertidal seascape (Alongi 2002) which interact to produce characteristic distributional ranges of most of the species (Duke et al. 1998). These may include salinity, soil type and chemistry, nutrient content, physiological tolerances, predation and competition (Smith 1992; Semeniuk 1994; Saenger 2002). Of these many factors, however, salinity and tidal inundation have the greatest impact in defining mangrove zones. According to Ball (1988), interspecific differences in salt tolerance might have contributed to the segregation of species along a salinity gradient. Varying frequency and duration of submergence attributed to tidal inundation creates gradients in several physical variables to which mangroves respond (Hogarth 1999).

Function and Uses of Mangroves

The mangrove ecosystem is a valuable economic and ecological resource (Naylor and Drew 1998). Mangroves are an important source of plant and wood products. They are the closest source of firewood to local coastal communities. Poles used in fishing gear and planks used for making boats and their accessories are usually obtained from the mangroves. They also provide wood products for housing and building constructions. The bark is also used for tanning fishing nets. Some of the other goods obtained from the mangrove trees include charcoal, dyes, glue, oil, food and drinks, medicine, paper, woodchips for conversion into rayon, fibres, ropes, corks and a lot more (Field 1995; Bandaranayake 1998; Ewel et al. 1998; Hogarth 1999; Saenger 2002; Primavera et al. 2004).

Mangroves have been utilized both in a sustainable and destructive manner. Mangrove waterways are important sites for small-scale cultivation of shellfish, finfish and crustaceans (Alongi 2002). The waterways also provide a means of communication and transportation between coastal settlements where there are no roads (Saenger 2002). Lately, mangroves are becoming a recreational and educational area, with boardwalks constructed to allow access into the forest (Primavera et al. 2004). Ecotourism has been popular in the mangroves where tourists come to see the scarlet ibises *Eudocinus ruber* in Trinidad, go kayaking among the mangroves in Honduras, for bird watching in the day and firefly viewing at night in Malaysia (Hogarth 1999), for viewing wildlife such as wading birds and alligators in Florida (Ewel et al. 1998), and for boating trips in the pristine mangroves of Palawan Island in the Philippines. One of the most destructive ways of utilizing mangroves is conversion to other land use such as ports, dams, agriculture lands, salt beds, industrial areas and aquaculture ponds (Primavera 1997; Alongi 2002; Saenger 2002).

In the Philippines, mangroves have been utilized in the past in naming sites of villages, towns and cities (Primavera et al. 2004). The capital city of the country was named after *Scyphiphora hydrophyllacea*, locally known as *nilad*. This species was common along Pasig River where the Spaniards under Miguel Lopez de Legazpi sailed. The locality lying along the river's deltaic plains was then called Maynilad, which was subsequently shortened to Manila (Saenger 2002).

Aside from the many goods and uses they provide mankind, mangroves have important ecological functions. Mangrove communities protect shorelines during storms and typhoons by absorbing wave energy and reducing water velocity passing through their intricate, entangled above-ground root system (Mazda et al. 1997). The extensive cable root system of some species assists in binding sediment particles, thereby facilitating accretion and preventing coastal erosion (Hogarth 1999). Riverine mangroves, on the other hand, reduce water velocity by adding flood storage capacity, thus mitigating flooding (Ewel et al. 1998). Although claims of Kathiresan and Rajendran (2005) that mangroves protected coastal residents from the December 2004 tsunami were questioned by Kerr et al. (2006); Danielsen et al. (2005) and Kar and Kar (2005) have also reported the mitigating effect of mangroves during the tsunami. Mangroves also maintain estuarine water quality. Suspended matter, nutrients and heavy metals are reduced by the mangrove root system (Clark et al. 1998). In areas where mangroves, seagrass beds and coral reefs occur adjacent to each other, the functional or ecological values of such systems are significantly enhanced (Fortes 1988; Mumby et al. 2006). The sediment and nutrient retention function of the mangroves provide waters suitable for seagrass and coral reef development (Wolanski et al. 1997).

Despite an undeserved reputation for being dull and homogenous systems, mangroves have highly complex patterns and processes that plays a key role for many commercially important species which spend their larval stages in mangroves, but spend adult life as benthic, pelagic or demersal species (Bridgewater and Cresswell 1999). Mangroves function as a nursery habitat and its interdependence with seagrass beds and coral reefs is apparent in the movement of fish and other organisms observed between these three adjacent systems (Gillanders et al. 2003; Sheridan and Hays 2003; Mumby et al. 2006). As habitat for a range of organisms, both from the terrestrial and the marine environment, mangroves themselves are unique systems with very high biodiversity (Macnae 1968; Alongi 2002).

Mangrove-associated fauna

Mangroves provide refuge from predators (Ronnback et al. 1999) and protection against desiccation (Fondo and Martens 1998), making a suitable habitat for a diverse

range of organisms, including zooplankton, sponges and ascidians, crustaceans such as prawns, shrimps, lobsters and crabs, insects, molluscs, fish, amphibians, reptiles, birds and mammals, as well as diverse benthic infauna (Kathiresan and Bingham 2001). Macnae (1968) divided the mangrove fauna of the Indo-West-Pacific into terrestrial and marine animals. The latter group included vertical dwelling animals that live on the trees and horizontal species which were the majority of the animals. Macintosh (1988) reviewed the ecology and physiology of mangrove decapods, emphasizing the species that are important to fisheries and aquaculture.

Terrestrial animals that dwell in mangroves include insects, termites, ants, mosquitoes, sandflies, midges, fireflies, spiders, amphibians, reptiles, birds, and mammals (Macnae 1968; Ewel et al. 1998; Hogarth 1999; Kathiresan and Bingham 2001; Saenger 2002). Veenakumari et al. (1997) reported 276 species of insects in the mangroves of Andaman and Nicobar Islands. In Australia, 16 ant species were recorded by Clay and Andersen (1996). Mangroves also provide an important habitat for landbirds, shorebirds and waterfowl which may either be permanent residents that forage and nest in the mangroves, some of which exhibit site fidelity, or temporary visitors (Kathiresan and Bingham 2001). In the Sunderbans of Bangladesh, Hussain and Acharya (1994) reported some 35 species of reptiles to include crocodiles, lizards, snakes and turtles and four genera of frogs. Among the mangrove mammals reported by Macnae (1968), Ewel et al. (1998) and Kathiresan and Bingham (2001) are tigers, monkeys, wild pigs, deer, flying foxes, otters, rabbits, rats, and the now locally extinct rhinoceros, wild buffalo, swamp deer and hog deer in the Sunderbans (Hussain and Acharya 1994).

Aside from zooplankton, sponges, ascidians, epibenthos, infauna and meiofauna, most of the marine species associated with mangroves, such as the crustaceans, molluscs and fish are important fisheries resources. According to Macnae and Kalk (1962), distribution of these animals in the mangroves is determined by the level of the water table, their resistance to water loss and their demand for protection from the sun, the degree of consolidation of the substrate and availability of suitable food.

Fish fauna in mangroves is highly diverse. The Embley River estuary in Australia, alone is home to some 197 species of fish (Kathiresan and Bingham 2001). In a

mangrove estuary in the southwestern of Thailand, 134 species of fish belonging to 49 families have been identified by Tongnunui et al. (2002). In the mangroves of Pagbilao, Quezon, Philippines, Pinto (1987) reported some 128 fish species belonging to 54 families, with *Ambassis kopsi* as the most abundant species. Twelve years later, *Ambassis kopsi*, together with *Ambassis urotaenia* and *Atherinomorus balabacensis* was still the most abundant fish species in the area (Ronnback et al. 1999). In the mangroves of Selangor in Malaysia, 119 species of fish were recorded in inlets and creeks while 37 species were caught in traps set on the mud flats (Sasekumar et al. 1992). In Florida, aside from other smaller fish species, mangroves are also being inhabited by large voracious predators such as the lemon shark *Negaprion brevirostris* (Franks and Gruber 2006) and the barracuda *Sphyraena barracuda* (Serafy et al. 2006).

Molluscs and crustaceans are among the most important fisheries resources from the mangroves, and also contribute to the high biodiversity of the ecosystem. In a study in the Semantan mangrove forest in Sarawak, Malaysia, 44 mollusc and 31 crab species were recorded by Ashton et al. (2003). In an earlier study conducted in Kapar Mangrove Forest Reserve in Selangor, Malaysia, Sasekumar (1974) recorded 26 gastropods, 1 teridinid and 48 species of crustaceans, predominantly brachyuran crabs. Among the molluscs found in mangroves, bivalves are the more economically important group. These include oysters (*Crassostrea* spp.) (Hogarth 1999), corbiculids (*Geloina* spp.) (Kathiresan and Bingham 2001), cockle (*Anadara* spp.) (Saenger 2002), and lucinid clams such as *Lucina pectinata* (Frenkiel et al. 1996) and *Anodontia edentula* (Lebata 2000). Both lucinid clams harbour endosymbiotic sulphur-oxidizing bacteria which allow them to thrive deep in sulphide-rich mangrove mud. *Anodontia edentula* is one of the most expensive and sought after bivalves in the Philippines and is extensively harvested in northern Iloilo in Panay Island (Lebata and Primavera 2001) and in Guimaras Island (Primavera et al. 2002). Another local mollusc delicacy, popular in the coastal areas in the Philippines, is the shipworm (Family Teredinidae) which bores into decaying wood of mangrove trees (Primavera et al. 2004).

Mangrove habitats and shrimp populations are tightly linked (Kathiresan and Bingham 2001) as shown by Vance et al. (1990) and Vance et al. (1996) in Australia,

Primavera (1998) in the Philippines, and Mohan et al. (1995) in India. Most of these shrimps are penaeids of the genus *Penaeus* (unrevised) and *Metapenaeus*. According to Kathiresan and Bingham (2001), juveniles of eight penaeid prawn species, primarily *Fenneropenaeus* (as *Penaeus*) *indicus* and *Metapenaeus monoceras*, are common in the Pichavaram mangroves in India. In Selangor, Malaysia, 9 penaeid species were caught in inlets and creeks while 11 species from the adjacent mud flats (Sasekumar et al. 1992). In Quezon, Philippines, Ronnback et al. (1999) observed that among the shrimps, the most common were penaeids, palaemonids, and *Acetes* sp. Another crustacean that is closely associated with mangroves is the mud lobster *Thalassina anomala* which can cause a profound impact on the topography and vegetation structure of the mangrove forest (Macintosh 1988). Crabs are also characteristic members of the invertebrate mangrove fauna. The most abundant mangrove crab species in the world belong to families Grapsidae and Ocypodidae (Hogarth 1999). Among the grapsid crabs, the most important in terms of diversity and abundance are the members of the genus *Sesarma*, while the fiddler crabs *Uca* spp. are the most important ocypodids (Macintosh 1988; Hogarth 1999; Kathiresan and Bingham 2001). Mangrove-associated portunids which are important fisheries resources include *Scylla* spp. (Walton et al. 2006a), *Portunus* spp. (Cooper 1997; Manson et al. 2005), *Charybdis* spp. (Dineen et al. 2001) and *Callinectes sapidus* (Yeager et al. 2006). Among these crabs, *Scylla* spp. and other less commercially important crabs such as the *Thalamita crenata*, *Baptozius vinosus*, *Episesarma versicolor* and *Cardisoma carniflex* have been collected in the natural mangroves where the present study was conducted and in the nearby replanted mangroves (Langdown 2005; Tapper 2005; Walton 2006).

Of the fisheries resources derived from the mangroves, in the Philippines and throughout Southeast Asia, the mud crab *Scylla* spp. is one of the most economically important products (Escritor 1972; Overton et al. 1997; Carpenter and Niem 1998; Kosuge 2001; Marte 2003). It is also an important mangrove commodity in Australia (Fielder and Heasman 1978; Le Vay 2001) and in Africa, where it is a prey of choice for women and children gleaning in Inhaca, Mozambique because it is valued highest for its taste, monetary value and household importance (de Boer et al. 2002). According to Macintosh (1988), *Scylla* is the only mangrove-associated portunid crab

that commands considerable interest because of its importance to fisheries and aquaculture.

THE MUD CRABS, *SCYLLA* SPP.

The mud crabs of the genus *Scylla* are large, edible crustaceans associated with mangrove swamps and have been observed throughout the Indo-West Pacific region (Macnae 1968; Carpenter and Niem 1998). These crabs substantially support small-scale fisheries in the coastal zone (Quinn and Kojis 1987; Prasad and Neelakantan 1988) and are harvested in Australia (Fielder and Heasman 1978), Papua New Guinea (Quinn and Kojis 1987), Malaysia (Kosuge 2001), Singapore (Chua 1973), Philippines (Estampador 1949a; Escritor 1972; Walton et al. 2006a), Thailand (Overton and Macintosh 2002), Vietnam (Le Vay et al. 2001), Japan (Imai et al. 2004; Imai and Takeda 2005), Sri-Lanka (Jayamanne and Jinadasa 1991), Mozambique (de Boer et al. 2002), and South Africa (Robertson and Piper 1991; Fratini and Vannini 2002).

Knowledge of the biology, distribution and population structure of a species is important for the development of proper fisheries management strategies. Thus, establishing the correct identification of a species is basic before commencing any experiment or research on any particular organism.

Morphology

The morphology of *Scylla* (Fig. 1.2), as described here has been adapted from Estampador (1949a) and Keenan et al. (1998). The carapace is oval, wider than long, moderately convex and has a smooth surface. The front is divided into 4 teeth ranging from low, rounded lobes to prominent, sharp spines and the frontal width varies between species. The antero-lateral margin is convex with 9 teeth of almost equal size. The infra-orbital margins are prominently toothed and the orbits exhibit no dorsal inclination. The antennules are folded transversely with a basal antennal article produced into the orbit; the flagellum lies in orbital hiatus.

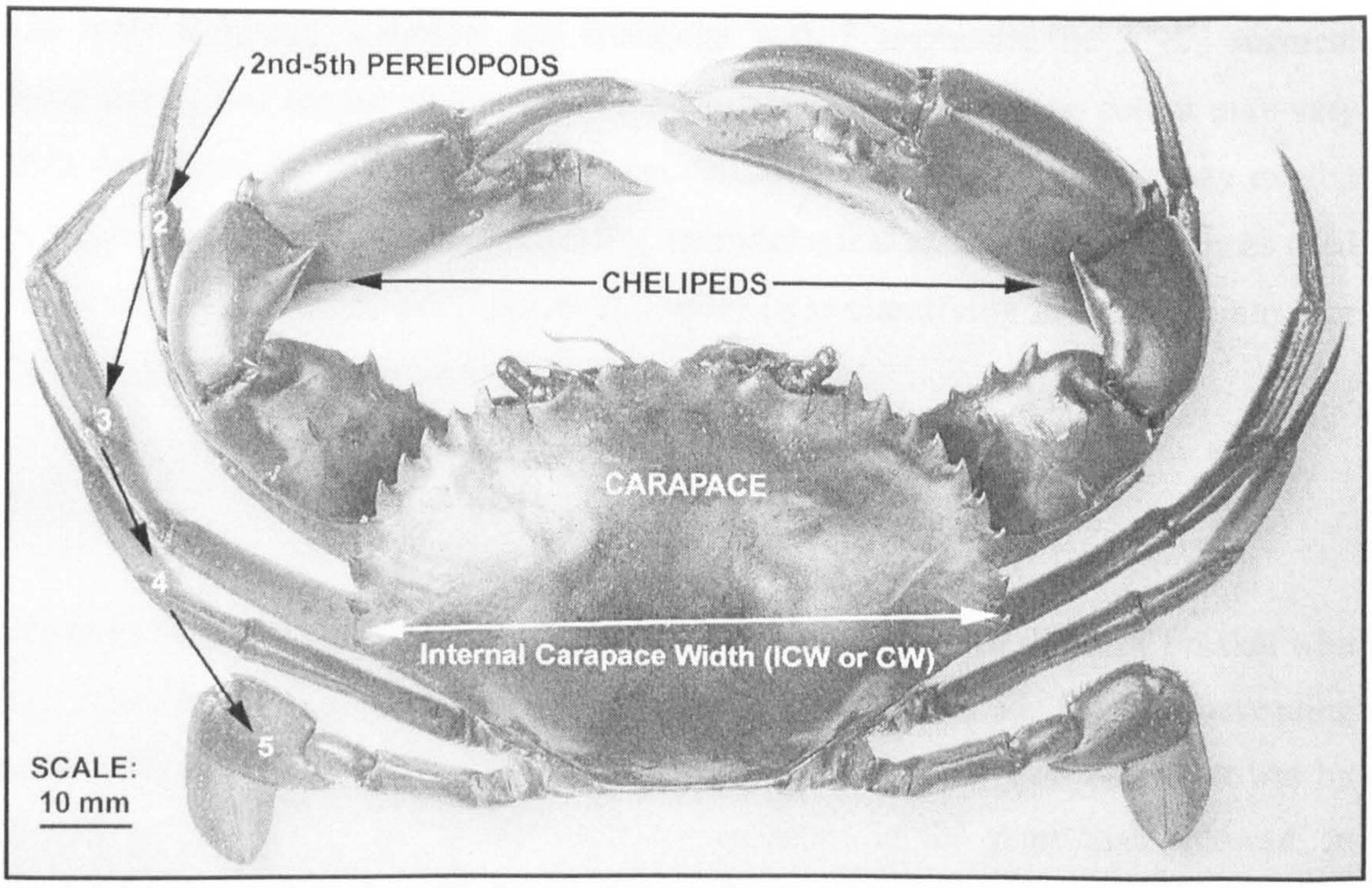

Figure 1.2. Adult *Scylla olivacea* showing the morphology of the mud crab belonging to the genus *Scylla*. The carapace width (CW) of crabs measured and used in this study is the internal carapace width (ICW) which is the distance between the 8th and 9th antero-lateral spines of the two sides of the carapace.

The chelipeds are massive, smooth and longer than any of the legs. The arm, wrist and hand have characteristically placed spines, with the 'hands' described as deep and full. The remaining legs (pereiopods 2-5) are stout and moderately compressed. Pereiopods 2-4 are similar and used for walking while each of the fifth pair has a shortened and flattened merus and carpus adapted for swimming. In females, the pleopods are used for attaching eggs which are carried by the female until hatching (Hartnoll and Gould 1988).

The male abdomen is narrow and triangular with 5 segments, the 3rd-5th segment being fused. The female abdomen is broad and U-shaped. Carapace colour may vary from dark brown to purple to light green. Walking and swimming legs may exhibit polygonal patterns. Polygonal patterning, morphological and genetic differences used by Keenan et al. (1998) and Imai et al. (2004) in re-classifying *Scylla* spp. into four distinct species are shown in Table 1.1.

Taxonomy

Confusion in the species identification of *Scylla* started after the death of Forskål who first identified *S. serrata* as *Cancer serratus* (Keenan et al. 1998). Succeeding taxonomists were not able to find the type specimen from the Red Sea described by Forskål in 1775. This led to nomenclature problems in the years that followed. In 1798, Fabricius described *Portunus tranquebaricus*. However, the specimens used were re-examined by Keenan et al. (1998) and found to represent all four species currently recognized. In Southeast Asia, where usually all four species occur, identification may be confusing at times but this is simplified in areas where only one species occurs; the Red Sea, South Africa and East Africa are exclusively *S. serrata* territories (Keenan et al. 1998).

Table 1.2 shows the chronological order of how the four *Scylla* species arrived to the present re-classification (Estampador 1949a; Stephenson and Campbell 1960; Keenan et al. 1998; Keenan 1999; Ng and Ahyong 2001; Imai et al. 2004). Following Keenan et al. (1998) and ITIS (2006), the taxonomic hierarchy and nomenclature of the four *Scylla* species are as follows:

Table 1.1. Morphological and genetic differences in the four *Scylla* spp. as described by Keenan et al. (1998) and Imai et al. (2004). The last column shows the number of base pairs (bp) in each haplotype from amplified 16S rDNA markers of each species.

Species	Shape of frontal lobe spines	Height of frontal lobe spines	Carpus spines of the cheliped	Propodus spines of the cheliped	Polygonal patterns on the legs	16S rDNA haplotype
Scylla serrata	blunt point	high	both obvious	obvious	prominent on all legs	2 (268, 268 and 55 bp)
Scylla tranquebarica	blunt	moderate	both obvious	obvious	prominent only on back pair of legs	1 (325 and 238 bp)
Scylla olivacea	rounded	low	inner absent, outer reduced	reduced	absent	3 (238, 188 and 111 bp)
Scylla paramamosain	triangular	moderately high	inner absent, outer reduced	obvious	weak	4 (277 and 238 bp) and 5 (277, 174 and 64 bp)

Kingdom: Animalia

Phylum: Arthropoda

Subphylum: Crustacea

Class: Malacostraca

Subclass: Eumalacostraca

Superorder: Eucarida

Order: Decapoda

Suborder: Pleocyemata

Infraorder: Brachyura

Superfamily: Portunoidea

Family: Portunidae

Genus: *Scylla*

Species:

Scylla serrata (Forskål, 1775)

Scylla paramamosain (Estampador, 1949)

Scylla olivacea (Herbst, 1796) (shown in Fig. 1.2)

Scylla tranquebarica (Fabricius, 1798)

Prior to the re-classification of the genus *Scylla* by Keenan et al. (1998), identification of all mud crabs as *S. serrata* was not uncommon as it had long been assumed that only one species existed. Macintosh (1988) and Overton et al. (1997) then considered *S. serrata* as the only species of family Portunidae that is closely associated with mangrove environments. Estampador (1949a) recognized four distinct groupings of mud crabs based on colour patterns, relative size, cheliped spination, chromosome form and the process of gamete development. However, the work of Stephenson and Campbell (1960) placing all morphs in synonymy under *S. serrata* became more widely accepted. Overton et al.'s (1997) canonical variate analysis of the morphometric and meristic characters of the species from Malaysia and Thailand contradicted this view; it showed three phenotypically different groups of crabs. This led them to suggest that the taxonomy and biology of *S. serrata* merited further study. Following Overton et al. (1997), Keenan et al. (1998) assessed genetic differences between mud crab specimens employing the use of allozyme electrophoresis and mitochondrial DNA sequencing followed by the use of discriminant function analysis

Table 1.2. Chronology of the development of taxonomic classification of *Scylla* spp.

Author & Year	SPECIES			
Forskål 1775	*Cancer serratus*			
Herbst 1796			*Cancer olivaceous*	
Fabricius 1798		*Portunus tranquebaricus*		
Ruppell 1830	*Portunus serratus*			
De Haan 1833	*Portunus (Scylla) serratus*			
H. Milne Edwards 1834		*Lupa lobifrons*		
MacLeay 1838	*Achelous crassimanus*			
Dana 1852	*Scylla tranquebarica* var. *oceanica*			
Stebbing 1910	*Achelous crassimanus*			
Estampador 1949	*Scylla oceanica*	*Scylla tranquebarica*	*Scylla serrata*	*Scylla serrata* var. *paramamosain*
Barnard 1950	*Scylla serrata*			
Serene 1952	*Scylla serrata* var. *paramamosain*	*Scylla tranquebarica*	*Scylla serrata*	*Scylla oceanica*
Crosnier 1962	*Scylla serrata*			
Guinot 1967	*Scylla serrata*			
Holthuis 1978				*Scylla serrata*
Joel and Raj 1980	*Scylla tranquebarica*		*Scylla serrata*	
Melo 1983	*Scylla serrata*			
Keenan et al. 1998	*Scylla serrata*	*Scylla tranquebarica*	*Scylla olivacea*	*Scylla paramamosain*
Imai et al. 2004	*Scylla serrata*	*Scylla tranquebarica*	*Scylla olivacea*	*Scylla paramamosain*

in determining morphologically defining characters for each of the genetically defined groups and the results showed four distinct species of *Scylla*. Yet, despite the recent developments on the species identification of *Scylla* (Keenan et al. 1998), some authors like Ronquillo et al. (1999), Fushimi and Watanabe (1999) and Klinbunga et al. (2000) preferred to follow the classification of Estampador (1949a). However, Keenan et al.'s (1998) work, is further supported by the recent findings of Imai et al. (2004) identifying all four species of *Scylla* using genetic markers of nuclear and mitochondrial DNA both from broodstock and larvae (Table 1.1). Moreover, Imai and Takeda (2005) reported a *Scylla* hybrid, possessing morphological and genetic characteristics of *S. serrata* and *S. olivacea*. This was assumed to be caused by the increased frequency of contact among different species of mud crabs due to the expansion of distributional ranges of several species of *Scylla* in Japan. This expansion was also observed in Australia by Gopurenko et al. (2003).

The re-classification of *Scylla* by Keenan et al. (1998) has major implications both for fisheries and aquaculture of mud crabs. The accuracy of works prior to 1998 may be in doubt due to reservations on the identity of the species. This means that more work needs to be done on the re-classified species of *Scylla* in order to understand better the biology and environmental requirements of each species which is essential for appropriate resource management.

Distribution

Scylla spp. have always been linked to mangroves in the Indo-Pacific (Arriola 1940; Estampador 1949a; Macnae 1968; Sasekumar 1974). Figure 1.3 shows the distribution of *Scylla* spp. adapted from FAO (2006). Comparing it with Figure 1.1, an overlapping distribution of *Scylla* and mangroves, except in South America, can be observed. Despite this knowledge of their habitat, very few research studies have been conducted for confirmed individual species of *Scylla* due to problems in species identification which were resolved only in the recent years. All four species of *Scylla* follow the same general life cycle, have overlapping distributions and share similar morphological features (Moser et al. 2005), hence the confusion in species identification prior to Keenan et al.'s (1998) work.

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Figure 1.3. *Scylla* spp. distribution shown in blackened areas. (Adapted from FAO 2006. Fisheries Global Information System. Food and Agriculture Organization of the United Nations, Rome, Italy).

Scylla serrata is the most widespread, occurring naturally throughout the Indo-Pacific. Although it can tolerate reduced salinity, it is commonly associated with mangrove forests inundated for most part of the year with full salinity oceanic water and is dominant in oceans where salinity is greater than 34 p.s.u.. They can be found in the Red Sea, South Africa, Mauritius and Western Australia in the Indian Ocean, in Northern Territory, Gulf of Carpentaria and Indonesia in the Arafura Sea, and in Fiji, Solomon Islands, New Caledonia, Western Samoa, Philippines, Japan, Taiwan and the east coast of Australia in the Pacific Ocean. It is sympatric with *S. olivacea* in the Matang mangrove forest in Malaysia (Kosuge 2001), and in the Gulf of Carpentaria, Western Australia, Philippines and Indonesia (Keenan 1999). Two clades of *S. serrata* were reported by Gopurenko et al. (1999); one that is strictly confined to northern Australia and the other that is widespread throughout the Indo-West Pacific. Moreover, Fratini and Vannini (2002) supported this result by confirming the occurrence of its population within a larger area of the Indian Ocean region. Gopurenko et al. (2003) found that *S. serrata* has recently become established in some south-west Australian estuaries, about 1000 km south of its recorded distribution.

Scylla tranquebarica, on the other hand, is commonly found in the South China Sea and in some specific locations in the Indo-Pacific. It is commonly found in mangrove forests and coastlines inundated with reduced salinity for part of the year. They have been collected from Pakistan and Malaysia in the Indian Ocean, from the Philippines in the Pacific Ocean, and from Malaysia and Singapore in the South China Sea (Keenan et al. 1998). They are sympatric with *S. olivacea* in Malaysia (Ikhwanuddin and Oakley 1999), Philippines and Singapore and appear to have a centralized distribution in the South China Sea (Keenan 1999). Although second to *S. olivacea* as the most common in the markets in Southeast Asia, *S. tranquebarica* is less common in Thailand and the Philippines (Carpenter and Niem 1998).

Scylla olivacea, though commonly found in the South China Sea, also occurs in specific locations in the Indo-West Pacific. It is very common in mangrove forests and coastlines inundated with reduced salinity seawater especially during wet season. It is common in areas where salinity falls below 31 p.s.u. and is limited to few embayments where salinity is reduced for extended periods during the monsoon season. Samples have been collected from Western Australia, Thailand and Pakistan

in the Indian Ocean, from the Philippines in the Pacific Ocean, from Thailand, Singapore, Vietnam, Malaysia, China, Taiwan in the South China Sea, and from Indonesia and Gulf of Carpentaria in the Arafura Sea (Keenan et al. 1998). It is very common in Southeast Asia and appeared to have a centralized distribution in the South China Sea where of Keenan's (1999) collection, it has a strong representation in the samples from the Philippines and Malaysia. It has a strong preference for the intertidal zone in Thailand (Moser et al. 2005) and is usually encountered in a limited area in the vicinity of river mouths or the innermost parts of the bays in Japan (Imai et al. 2004). Together with *S. paramamosain*, it dominated the mangroves in Ban Don Bay, Thailand (Overton and Macintosh 2002). Dudley (2001) reported that crab landings in Segara Anakan, Indonesia were comprised of all four species of *Scylla*, mainly dominated by *S. olivacea*. This was also observed in the present study where *S. olivacea* is sympatric with *S. serrata* and *S. tranquebarica*.

Scylla paramamosain is abundant in the continental coast of the South China Sea and south into the Java Sea. Its association varies from coral reef rubble in Singapore to shallow subtidal and estuarine ponds in Central Java to the mangrove forests in Mekong Delta in Vietnam (Keenan et al. 1998; Le Vay et al. 2001; Ut 2002; Walton et al. 2006b). It has been also positively identified in the South China Sea encompassing China, Hong Kong, Taiwan, Singapore and Cambodia, and in the Java Sea (Keenan et al. 1998; Keenan 1999). It is sympatric with *S. olivacea* and *S. serrata* in Japan (Imai et al. 2004; Takano et al. 2005) and with *S. olivacea* in Vietnam (Le Vay et al. 2001; Walton et al. 2006b).

Although closely associated with mangroves, *Scylla* species are not restricted to this habitat (Robertson 1996). They have been reported to inhabit algal and seagrass beds adjacent to the mangroves of Picharvaram in India (Chandrasekaran and Natarajan 1994), and in coral reef rubble in Singapore (Keenan et al. 1998). Spawning female *S. serrata* have been caught up to 95 km from the shore, at depths of 60 m in South Africa (Hill 1994). Aside from the availability of a suitable habitat, mud crab distribution may be determined by salinity, temperature, and the availability of food and the associated competition and predation factors.

Life cycle

The life cycle of mud crabs is divided into 4 stages, larvae, juveniles (20-70 mm carapace width or CW), sub-adults (70-150 mm CW) and adults (150 mm CW or above). Mud crab development starts with the mating of a hard-shelled male and a soft-shelled female, typical of the portunids (Hartnoll 1969). This usually takes place within 3 days after moulting, when the shell of the female is still hardening (Ong 1966). Although mating usually takes place inside the burrows (Nandi and Dev Roy 1990, see next section), Robertson and Kruger (1994) also reported catching copulating pairs or pairs in post-copulatory embrace in traps. Once deposited into the female, sperm in the spermatophores remains viable for more than 5 months, which explains why berried crabs under laboratory conditions were able to undergo successive spawnings without mating (Ong 1966). According to Ronquillo et al. (1999), during extrusion, sperm from the seminal receptacle mix with the eggs but gamete activation and fertilization only take place in seawater which may be the reason why after mating, mature females migrate into the open sea where they extrude their eggs (Arriola 1940).

Spawning is usually accompanied by migration to the open sea where temperature and salinity may be more favourable for the larvae (Hill 1974; Hill 1994) and which may facilitate larval dispersal (Hill 1994). Perrine (1979) reported that spawning migrations were stimulated by decreases in salinity. Arriola (1940) and Estampador (1949b) also reported peaks of spawning activity in the Philippines from May to October which coincides with the rainy season. This is in agreement with Heasman et al. (1985), who associated spawning migration with freshwater inputs. Moreover, seasonal changes in salinity have been reported by Walton et al. (2006b) as an important factor in relation to recruitment of *S. paramamosain*. Berried *Scylla* spp. captured nearshore suggesting migration from the mangrove estuarine habitat to oceanic or near oceanic waters for spawning had been reported in the Philippines (Arriola 1940), Malaysia (Ong 1966), South Africa (Hill 1975; Robertson and Kruger 1994), Ponape, Caroline Islands (Perrine 1979), and Australia (Hyland et al. 1984; Hill 1994).

During spawning, released eggs covered by a sticky secretion were moved by currents from the inhalant chamber into a basket-like structure formed by the endopodites and abdominal flap where they became attached to the endopodite setae of the pleopods forming a 'sponge' (Estampador 1949b). For *S. serrata*, incubation period was almost 30 d at 20°C and 30 p.s.u. (Heasman and Fielder 1983) and hatching of most eggs takes place over a 1 to 2 hour period (Hill 1974). Zoea production in the hatchery as reported by Quinitio et al. (2002) were 0.8-4 million for a 350-525 g *S. serrata*, 0.7-3 million for a 240-300 g *S. tranquebarica* and 0.4-2.7 million for a 360-465 g *S. olivacea*. The larvae undergo five stages of zoea lasting 2-5 days at each stage before attaining the megalopa stage (Heasman and Fielder 1983; Ronquillo et al. 1999; Quinitio et al. 2002).

The zoea 5 larvae developed into megalopae approximately 19 days after hatching (Ronquillo et al. 1999). In hatchery production, megalopae were nursed either in concrete tanks at 26-32 p.s.u. or in brackish water ponds and approximately 30 days from hatching, megalopae metamorphosed to crab 1 (Quinitio et al. 2002). Metamorphosis to the first crab stage may last from 7 to 12 days depending on environmental conditions (Heasman and Fielder 1983; Ronquillo et al. 1999). Crab 1 juveniles weighed 1-3 g body weight or BW (Quinitio et al. 2002). According to Arriola (1940), juveniles measuring 16-48 mm CW appeared in estuaries in the Philippines at the end of September. Hill et al. (1982) also reported predominant occurrence of juveniles (20-99 mm CW) in the intertidal zone where they use mangrove roots and pneumatophores as refuge. In Papua New Guinea, recruitment was reported to be continuous because of the nature of reproduction which is non-seasonal (Quinn and Kojis 1987). Juveniles develop into mature crabs after 16-18 moults within 338-523 days (Ong 1966).

Temperature, salinity, burrowing

Temperature tolerance of the species may vary between locations. In the subtropics, Heasman et al. (1985) reported that the height of spawning in *S. serrata* in Moreton Bay, Queensland was between 24 and 28°C. This coincides with the optimal temperature of 27°C suggested by Heasman and Fielder (1983) as required for successful incubation and larval development. However, contradictory results have

been reported by Hill (1974) who revealed that zoea larvae of *S. serrata* from South Africa were not tolerant to high temperatures; the optimum temperature appears to be 14 to 20°C. At a later stage, instar 2 of *S. serrata* grown for 18 days at 20, 25, 30 and 35°C had the highest survival (98%) at 25°C while lowest (36%) at 20°C (Ruscoe et al. 2004). Catchability of *S. serrata* in Deception Bay, Australia was higher in the warmer months from September to April (mean of 18.7 crabs on each occasion) than the cooler months from May to August (4.8 crabs) (Hill et al. 1982). This was explained by the result of a laboratory experiment which revealed that feeding, emergence and movement of *S. serrata* was optimum at 20-25°C and declined at 16°C (Hill 1980).

In the tropics, spawning coincides with the summer months, a period of peak productivity and peak abundance of food organisms available to spawners. According to Hill (1974) larvae from tropical areas may be more tolerant to high temperatures than their subtropical counterpart. *Scylla* spp. in Kerala, India survived for 3 h when exposed to 40°C. Although no temperature readings was reported, Chandrasekaran and Natarajan (1994) claimed that seasonal variations in water temperature did not seem to influence the distribution of *Scylla* spp. juveniles in Pichavaram mangroves in the southeast coast of India. It was suggested that mud crab eye stalks produce a chemical responsible for it to be more tolerant of high temperatures (Hamumante et al. 1980).

Hill (1974) suggested that *S. serrata* zoea are unsuited to estuarine conditions based on the results of his experiment on first stage zoea larvae obtained from ovigerous females collected from Kleinemond estuary in South Africa, where considerable mortality was observed when larvae were exposed to salinities below 17.5 p.s.u.. This salinity requirement may be the reason why newly hatched crabs had never been collected in rivers but only in river mouths, along shorelines and offshore (Arriola 1940). Based on the result of their experiment on instar 2, Ruscoe et al. (2004) recommended that *S. serrata* juveniles from the Northern Territory of Australia should be reared at approximately 10-25 p.s.u. in order to achieve maximal production. According to Manjulatha and Babu (1998), *Scylla oceanica* (re-identified as *S. serrata*) could survive sudden fluctuations in salinity, but prolonged exposure to lower salinities below 10 p.s.u. resulted in poor feeding and reduced growth. However,

the St. Lucia system in a South African estuarine lagoon which experiences wide salinity fluctuations is a natural habitat to a *S. serrata* population. Hill (1979a) reported that mud crabs survived a four-month period of low salinity (2 p.s.u.) in 1976, but their hypersalinity tolerance (60 p.s.u.) is too low to permit their existence in most of the system during periods of high salinity. This wide range of salinity tolerance enables adults to cope with the conditions in intertidal pools and hypersaline lagoons (Hill 1979a). This finding contradicts Keenan et al.'s (1998) report that *S. serrata* is the least tolerant to low salinities of all the four *Scylla* species. The effect of salinity on the survival and development of *S. olivacea* showed that development of zoea to megalopa had significantly higher survival at 30 p.s.u. compared with 28, 32 and 34 p.s.u., while from megalopa to the fifth crab stage survival was significantly higher at 16, 20 and 32 p.s.u. compared with 12, 36 and 40 p.s.u. (Jantrarotai et al. 2002). According to Ronquillo et al. (1999), the best survival of *S. serrata* (re-classified as *S. olivacea*) zoea was recorded at 33-34 p.s.u. with the rate of zoeal development not significantly different between 20-40 p.s.u. at 27°C. However, total mortality was observed in zoea 1 after 5 days at 15 p.s.u.. Moreover, in southeast India, where recruitment appears to be year-round, juvenile crabs are most abundant in February, months later after the heavy rains when the salinity increased to >20 p.s.u. (Chandrasekaran and Natarajan 1994).

Salinity was observed by Walton et al. (2006b) as an important factor in the settling of *S. paramamosain* in Vietnam. Recruitment peaks when small crabs (6 mm mean internal CW) were settling out on the mangrove fringe were from December to February, coinciding with a rise in salinity as the dry season begins. As mentioned earlier, Perrine (1979) suggested that offshore migration in a tropical population is stimulated by a rapid decrease in salinity. In the tropics where temperature is more constant and seasonality is less defined, salinity may be a more important cue to migration, offshore spawning and larval settlement rather than temperature.

The burrowing activity of mud crabs has been reported by Estampador (1949a), Macnae (1968), Macintosh (1988), and Nandi and Dev Roy (1990). According to Atkinson and Taylor (1988), the function of burrows include concealment from predators, protection from adverse environmental conditions, provision of territorial centre, provision of site for feeding, moulting, mating, egg incubation or juvenile

recruitment, provision of oxygenated water within the sediment. Estampador (1949a) described *S. olivacea* as hole-dwellers while *S. serrata* as free-ranging. Although Macnae (1968) reported that all sizes of *S. serrata* construct burrows in the intertidal zone, that was the time when mud crabs were considered only as *S. serrata*. Thus, the samples referred to by Macnae (1968) as *S. serrata* may actually be *S. olivacea*. However, Estampador's (1949a) classification of *Scylla* was not recognized by most authors, including Macnae (1968). According to Hill (1978), in a tracking study of *S. serrata* in Kowie estuary, South Africa, crabs live a free-ranging, non territorial existence, some individuals moving up or downstream. The capacity of *S. serrata* for thermo-tolerance (Hamumante et al. 1980) may have enabled it to live a free-ranging life style as compared with *S. olivacea* which prefers to live in burrows (Estampador 1949a) where temperatures are considerably more stable than the outside environment (Atkinson and Taylor 1988). Nandi and Dev Roy (1990) reported that burrows can provide a temperature of 28-30°C when the shore temperature can be as high as 40°C. Aside from protecting them from temperature and salinity extremes, it also protects them from aggression from other crabs (Macintosh 1988). The use of burrows for moulting has also been reported by Nandi and Dev Roy (1990) who observed cannibalistic behaviour of female *Scylla* spp. toward moulting males inside burrows, while males serve as guards outside burrows of moulting female crabs.

Growth

Understanding the pattern of growth of individuals is a fundamental requirement for the proper management and conservation of the population since many ecological processes that affect population dynamics of coastal species are size dependent (Lee et al. 2006). In crustaceans, like the mud crabs, growth takes place through periodic ecdysis or moulting, which is the shedding of the confining exoskeleton (Zou and Fingerman 1999). According to Nandi and Dev Roy (1990), moulting crabs are usually hiding in their burrows. Prior to moulting, white air spaces can be observed under the carapace, accompanied by the swelling of membranes between the legs and the body (Walton 2006). Then, a crack appears around the carapace starting from the gill chambers spreading towards the dorsal suture between the abdomen and posterior carapace. The crab emerges backwards through this crack during ecdysis and the newly moulted crab takes up water to expand the new soft shell, thus increasing its

size (Fielder and Heasman 1978). It has been observed that these soft shelled crabs are vulnerable to predation and cannibalism.

According to Hartnoll (1982), the time between moults is known as the intermoult period and the increase in size between one intermoult period and the next is called the moult increment. Growth in crustaceans has been quantified with several measurements of body size and weight and relationships between these characteristic data have been described by linear or nonlinear models (Anger and Moreira 1998). Most growth studies have been conducted in pond experiments (Baliao et al. 1981; Agbayani et al. 1990; Fortes 1999b; Trino et al. 1999; Rodriguez et al. 2003; Christensen et al. 2004), and most pond studies have generated a variety of growth rates in different species of *Scylla*. These growth rates are usually expressed in weight since in aquaculture, weight is equated to production. Table 1.3 shows the growth rates (length or weight per day) of *Scylla* spp. obtained from the wild, laboratory experiment or pond culture.

Several environmental variables control growth in crustaceans. Some of these are temperature and salinity, or the nutritional status, age, size, and even missing limbs of an individual (Hartnoll 1982). Aside from these factors, growth in ponds are sometimes affected by the presence or absence of shelter, and differences in stocking density and diet (Fortes 1999b; Trino et al. 1999; Rodriguez et al. 2003). Growth can also be hastened by adding 14 or 18 μg diethylstilbestrol per gram diet every 3 days (Wang and Li 1989 as cited in Ut 2002).

In the subtropics, reduction in growth during winter was reported on *S. serrata* in South Africa (Hill 1975). He further noted that growth was faster during the first 18 months of life, but was reduced after. This observation was again reported in Queensland, where growth of sub-adults was found to be occurring only during spring and summer (Hill et al. 1982), probably as a result of higher water temperatures. Also in Queensland, Lee (1992) noticed that growth of *S. serrata* ceases in winter, when temperatures drop below 20°C. In the tropics, higher water temperature may increase the crab growth rates thus decreasing time to achieve maturity (Fielder and Heasman 1978). According to Heasman and Fielder (1983) optimal growth for *S. serrata* is at

Table 1.3. Growth rates of *Scylla* spp. both in weight and in length from different systems (wild, cultured or grown in the laboratory).

Species	System	Growth rate	References
Scylla olivacea	wild	0.25 mm d^{-1}	Thomas et al. (1987)
Scylla olivacea	wild	0.33 mm d^{-1}	Moser et al. (2002)
Scylla olivacea	wild	0.33 mm d^{-1}	Walton (2006)
Scylla paramamosain	wild	0.67 mm d^{-1}	Walton et al. (2006b)
Scylla serrata	laboratory	0.22-0.39 mm d^{-1}	Catacutan (2002)
Scylla spp.	pond	0.6 mm d^{-1}	Baliao et al. (1981)
Scylla spp.	pond	1.62-2.28 g d^{-1}	Agbayani et al. (1990)
Scylla olivacea	pond	0.12-0.25 mm d^{-1}	Fortes (1999b)
Scylla olivacea	pond	0.69-1.28 g d^{-1}	Fortes (1999b)
Mixed *Scylla serrata* & *Scylla tranquebarica*	pond	1.1 mm d^{-1}	Trino et al. (1999)
Mixed *Scylla serrata* & *Scylla tranquebarica*	pond	2.6 g d^{-1} (female) 3.9 g d^{-1} (male)	Trino et al. (1999)
Scylla paramamosain	pond	0.56-0.87 mm d^{-1}	Ut (2002)
Scylla olivacea	pond	1.22-1.37 g d^{-1}	Rodriguez et al. (2003)
Mixed *Scylla olivacea* & *Scylla paramamosain*	pond	2 g d^{-1}	Christensen et al. (2004)

27°C. Considering a more constant and elevated temperatures in the tropics, it is expected that growth is not seasonal, unlike in the subtropics where growth ceases as the temperature drops. Given the same age, tropical species are likely to be bigger than their subtropical conspecifics.

Fisheries

Mud crabs are an important source of income for coastal fishing communities (Overton and Macintosh 2002). They have been exploited traditionally in artisanal fisheries targeting all size classes of crabs in areas where fishing in unregulated; seed crabs for pond culture, adult and sub-adult for fattening, for the soft-shelled industry or for food (Overton et al. 1997; Dat 1999; Le Vay et al. 2001). Varying levels of overfishing or reduced landings have been reported in countries like Australia (Mounsey 1989), Sri Lanka (Jayamanne 1992), Thailand (Tiensongrusmee and Pratoomchat 1999), and all throughout Southeast Asia (Overton et al. 1997) where mud crabs are considered a delicacy (Marte 2003). The increasing demand coupled with rising prices subject them to heavy fishing pressure (Fielder and Heasman 1978). Moreover, the expanding export of mud crab as an alternative for shrimp has led to intensified harvesting, thus further threatening the wild stocks (Quinitio et al. 2002). Important export markets are Malaysia, Singapore, Hong Kong, Taiwan, Japan and U.S.A. (Millamena et al. 2001).

Mud crab fisheries employ gears such as gill nets, trawls, and traps. In some areas, handpicking of crabs is also a fishing method, as well as using hooks and axes. However, the most frequently used gear are the baited traps or pots which are common in Australia (Lee 1992), Indonesia (Cholik and Hanafi 1992), Philippines (Arriola 1940; Walton et al. 2006a), Thailand (Tookwinas et al. 1992), Bangladesh (Khan and Alam 1992), Sri Lanka (Jayamanne 1992), and South Africa (Robertson 1989). According to Robertson (1989), baited traps are the most practical and widely used gear for catching crabs. The use of traps is one of the easiest and more convenient ways of collecting crabs and the reason why fishermen and researchers alike prefer the use of traps to collect mud crabs both for commercial and research use (Heasman et al. 1985; Robertson 1996; Moser et al. 2005; Walton et al. 2006a).

Mud crab landings from capture fisheries is shown in Figure 1.4 (FAO 2006). From 1950-1973, it can be observed that mud crabs were solely coming from Asia. From 1974 to 2003, Asia provided 90-99% of the total mud crab production. Contribution of the Philippines to capture fisheries started in 1970 and production peaked in 1992 at 7,788 mt, experiencing a drastic drop of 3,125 mt from 1996 to 1997. During this peak in production, the Philippines contributed 41% of total mud crab production. The declining mud crab landings from artisanal fisheries is supplemented by increasing production from aquaculture from 1997 to 2003 (Fig. 1.5). Mud crab aquaculture is usually conducted in ponds constructed in former mangrove areas, the natural habitat of wild mud crabs that support capture fisheries.

Aside from overfishing, the decline in mud crab production may be further explained by the loss of habitat. There are several ways of addressing the problem of declining mud crab population, including regulation of fishing effort, rehabilitation of mangrove habitats, mangrove-friendly aquaculture, and enhancement of wild crab stocks. In Sri Lanka, Jayamanne (1992) strongly recommended the prohibition of capture of immature crabs, educating fisherfolk and developing aquaculture for mud crabs. Unlike in Australia, where all berried mud crabs, females with internal CW (<13 cm and males with internal CW<12 cm are protected under the 1995 Mud Crab Fisheries Management Plan (Heasman and Fielder 1977), and in South Africa, where the minimum landing size is 14 cm external CW through the 1998 Marine Living Resources Act 1998 (Robertson and Kruger 1994), no such regulation on mud crab fishing exists in the Philippines. This makes the wild population vulnerable to uncontrolled fishing. This is also experienced in Vietnam, where fisheries even for small crabs (10-100 g) exist. These small crabs are used as seed for stocking ponds (Overton and Macintosh 1997; Le Vay et al. 2001). Restoration of a completely degraded habitat proved to be successful in the replanted mangroves in Kalibo, Aklan with mud crab yields as high as 411 kg $month^{-1}$ (Walton et al. 2006a). Blankenship and Leber (1995), on the other hand, have been suggesting stock enhancement as one of the three main approaches to replenishing depleted stocks and managing fishery yields, especially in areas where habitat is not completely degraded.

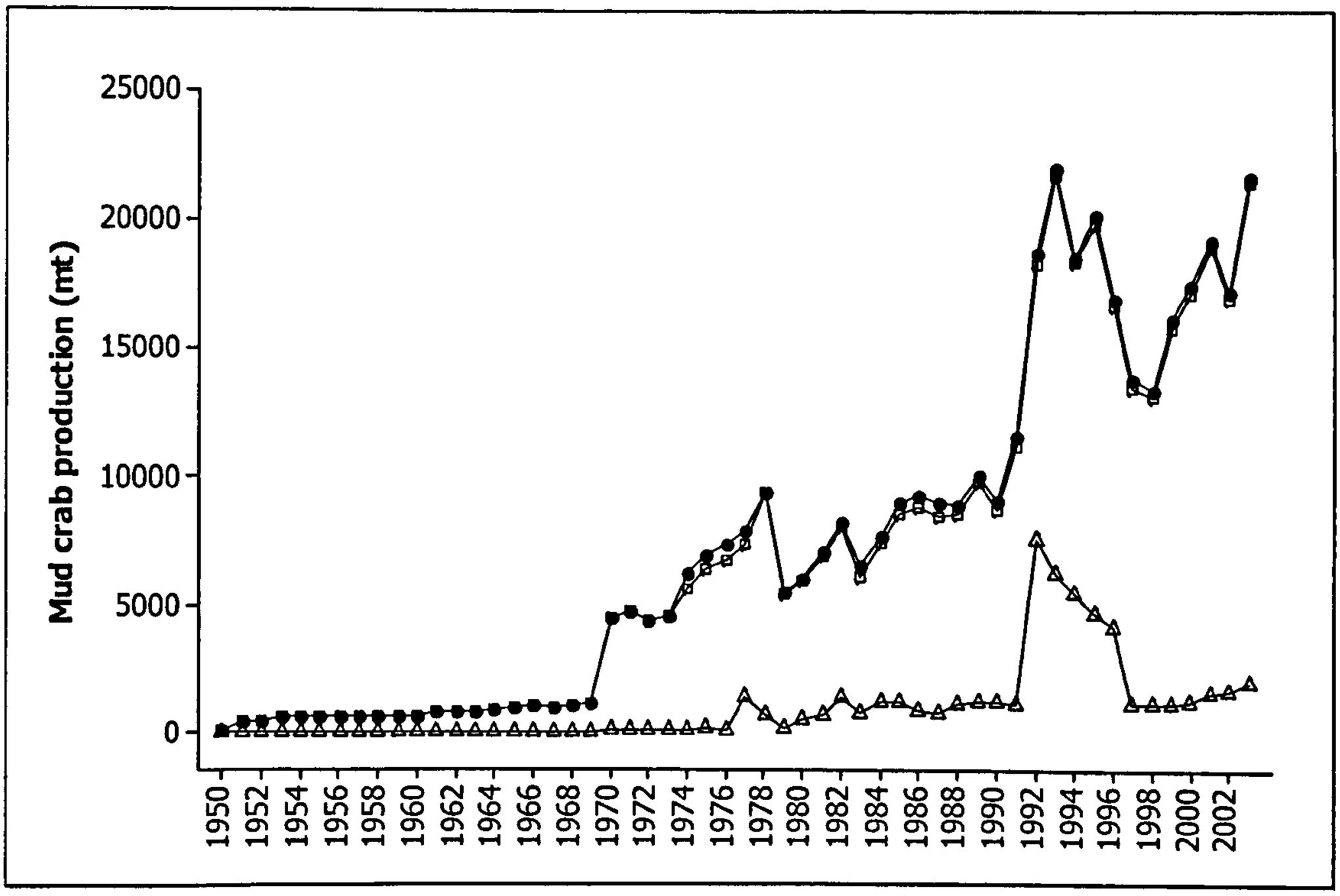


Figure 1.4. *Scylla* spp. production (1950-2003); total (filled circles), Asia (empty squares), and the Philippines (empty triangles). Values plotted were obtained from FAO 2006, Fisheries Global Information System, Food and Agriculture Organization of the United Nations, Rome, Italy.

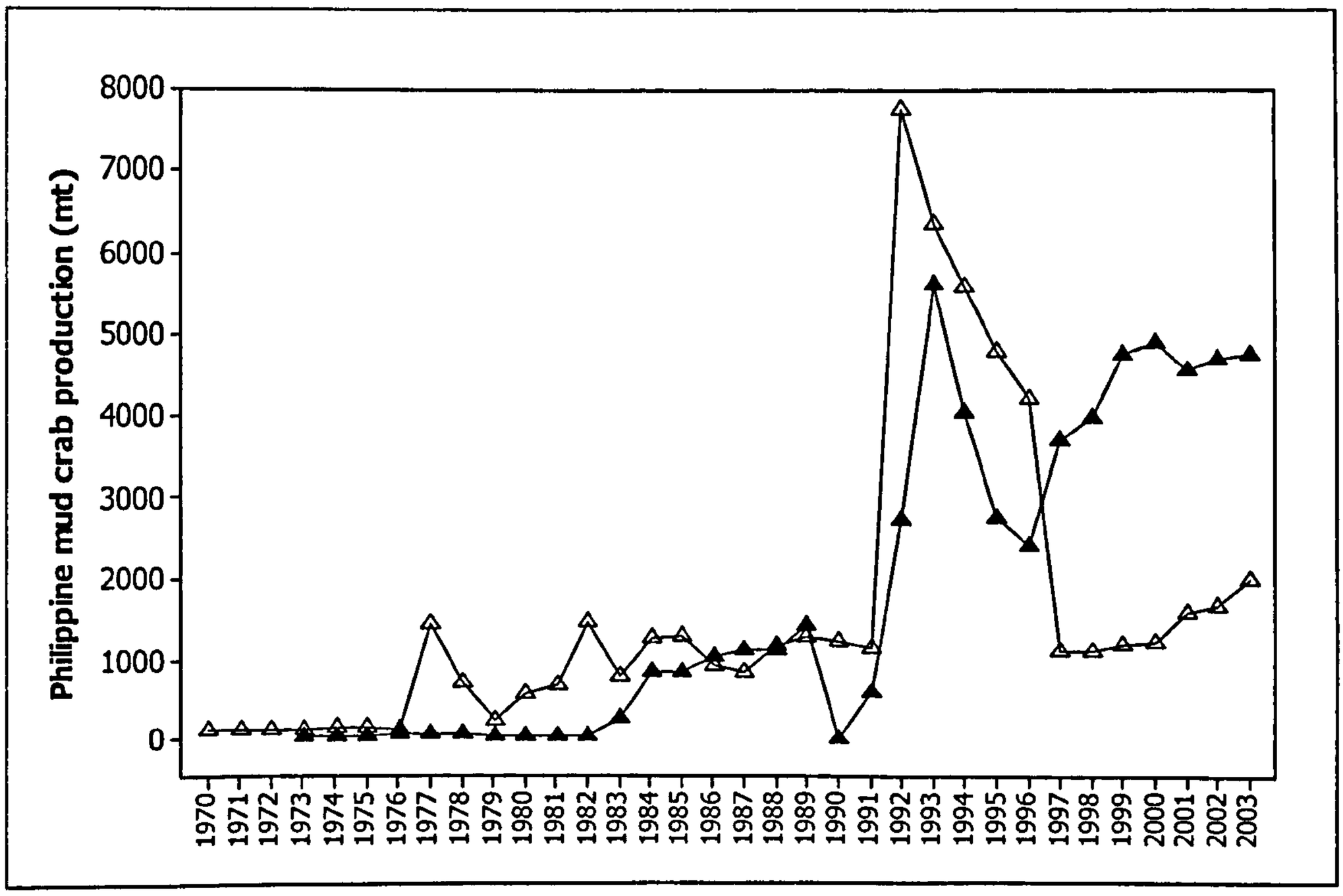


Figure 1.5. *Scylla* spp. production in the Philippines (1970-2003); capture fisheries (empty triangles), and aquaculture (filled triangles). Values plotted were obtained from FAO 2006, Fisheries Global Information System, Food and Agriculture Organization of the United Nations, Rome, Italy.

STOCK ENHANCEMENT

Background and Rationale

The problem of reduced productivity from many of the world's coastal and marine wild fisheries is caused by first, overfishing in all its forms (Pauly 1988) and second, the degradation of the ecosystems through coastal development and destructive fishing methods (Bell et al. 2005). The average annual landings of plaice *Pleuronectes platessa* by individual trawlers in the North Sea was clearly seen to be decreasing as early as 1880 with similar trends observed in haddock *Melanogrammus aeglefinus* and hake *Merluccius merluccius* (Gulland and Carroz 1968). The scallop *Patinopecten yessoensis* which was an important commodity for export in Japan since the 1600s peaked in 1920s in the Mutsu Bay and in 1930s in Hokkaido and started to drop from then (Ventilla 1982). The red abalone *Haliotis rufescens*, an important fishery resource in California started to decline in 1968 (Gaffney et al. 1996). The blue crab *Callinectes sapidus*, the most valuable fishery in Chesapeake Bay, also experienced some 55% drop from its record-high catches in 1993 to historically low levels in 2001 and 2002 (Zmora et al. 2005). To boost production, scientists and fisheries managers have been looking at ways of enhancing fish stocks for over a century (Blaxter 2000). Replenishing depleted stocks may be done first, by regulating fishing effort; second, by restoring degraded nursery and spawning habitats; or third, through stock enhancement (Blankenship and Leber 1995).

Stock enhancement using individuals reared in aquaculture facilities is becoming a popular method of supplementing depleted stocks (Bert et al. 2003). However, according to Bell and Nash (2004), the capability to produce and release juveniles from these aquaculture facilities is not enough reason to conduct stock enhancement. Stock enhancement success, according to Bell et al. (2005), depends on knowing enough about the ecology of the species, its nursery habitat, and the survival of cultured juveniles in the wild. This means gathering information about the population of the species concerned and its habitat prior to any attempts on stock enhancement. It has been stressed by Gulland and Carroz (1968) that the essential basis of any management is the proper biological understanding of the state of the stocks

concerned and emphasized that proper management requires good scientific knowledge based on adequate data. Blaxter (2000) further added appropriate size-at-release, season of release and area for release as keys to stock enhancement success. To ensure success, 10 components of a responsible approach to developing, evaluating and managing marine stock enhancement programmes have been prescribed by Blankenship and Leber (1995).

Hatchery-reared vs. wild

Although one way of enhancing stocks is through capture, culture and release of wild-caught juveniles (Bell et al. 2005), usually, juveniles used for stock enhancement, regardless of species, are produced in aquaculture facilities or hatcheries. However, several studies have documented defects or abnormalities both in the morphology and behaviour of hatchery-reared marine animals. These have been reported in fish, crustaceans and molluscs.

In cod *Gadus morhua* released in Scandinavia, hatchery-reared fish tended to eat sedentary prey such as molluscs while wild cod fed on fish and crustaceans (Blaxter 2000). Tilting behaviour of the red bream *Pagrus major* may effectively protect them against predation. However, non-tilting individuals which are highly vulnerable to predation appeared more frequently compared with tilting ones in intensive hatchery-reared juveniles (Uchida et al. 1993; Tsukamoto et al. 1999). Hatchery-reared milkfish *Chanos chanos* were observed to have an open operculum, shorter and stouter compared with that of their wild conspecifics (Hilomen-Garcia 1997). In the blue crab *Callinectes sapidus*, several differences between hatchery-reared and wild conspecifics have been noticed by Davis et al. (2005). Hatchery crabs raised in light blue fibreglass tanks which were less speckled and more uniform, were lighter in colour, greyer and a different hue than the wild crabs. They also tended to spend less time burying than their wild conspecifics. Hatchery crabs were also noticed to have shorter lateral spines than wild crabs. Colour, burying and lateral spines are important in avoiding predation. Mills et al. (2004) reported a promising behaviour of ongrown (wild-caught pueruli reared for one year in tanks) spiny lobster *Jasus edwardsii* which showed the same response as the wild lobsters when approached by divers, sensing the presence of predators. However, differences in diet selection in wild and ongrown

lobsters may have behavioural or morphological bases, such that prey items routinely eaten by the wild may not have been recognized as prey by ongrown lobsters. Higher stocking densities in the hatchery have also caused damage (loss of limbs and antennae, damage to swimming legs and swimmerets) to shrimps. Penaeid shrimps *Penaeus semisulcatus* stocked at a density ≥2,860 m^{-3} had significantly greater damage than those reared at 75 m^{-3} but which were capable of recovering from damage within 7 days once density is lowered to 75 m^{-3} (Heales et al. 1996). Survival of hatchery-reared *Penaeus esculentus* decreased with increasing stocking density when stocked in experimental raceway facilities in Brisbane, Australia (Loneragan et al. 2004). Variations in growth of giant clams *Tridacna* spp. which may be due to crowding in tanks had also been reported by Bell et al. (2005). Tank trials showed the lack of shelter-seeking behaviour of hatchery-reared red abalone *Haliotis rufescens* (Schiel and Welden 1987). Although the use of wild spat for stock enhancement of scallops showed no inherent morphological, behavioural or genetic deficits, Ventilla (1982) reported that ~10% of seeded *Patinopecten yessoensis* harvested from the seabed in Japan have deformed shells.

Based on these reports, hatchery-reared juveniles or hatchery-grown juveniles obtained from wild spat or pueruli may be less fit than their wild conspecifics. However, these inferiorities of hatchery-reared animals can be improved by employing different release strategies that may improve field survivorship and stock enhancement success.

Release strategies

Varying release strategies have been reported for different species depending on the nature of the release animals, their behaviour and their needs in the wild. Determining optimum size-at-release, conditioning animals prior to release, monitoring habitat prior to release, and even providing artificial shelters to released animals to protect them from predators have been tested, all with the same aim at increasing survival of released animals in the wild.

Growth rates and recovery rates had been positively correlated with size-at-release. Larger animals were observed to have better survival than smaller ones which are

more vulnerable to predation and cannibalism. A general trend of better survival with increasing size was observed in the topshell *Trochus niloticus* (Crowe et al. 2002), soft clam *Mya arenaria* (Beal and Kraus 1991), queen conch *Strombus gigas* (Ray et al. 1994), and turbot *Scophthalmus maximus* (Iglesias and Rodriguez-Ojea 1994; Stottrup et al. 1998). In the mullet *Mugil cephalus*, recovery rates were five times greater for fish released larger than 85 mm than those smaller than 60 mm total length (Leber 1995; Leber and Arce 1996). Acceptable rates of survival were also observed for scallops *Patinopecten yessoensis* released at ~30 mm shell length (SL) (Ventilla 1982). However, Ray et al. (1994) recognized that size-at-release for restocking is a trade-off between survival rates and production cost. Leber et al. (2005) also reported that cost production increases with increasing size of juvenile mullet *Mugil cephalus* with wages and benefits taking up the highest percentage of expenses and an increasing cost in feed as the fish grow.

Evidence of improved performance and survivorship of hatchery-reared animals through conditioning in the hatchery prior to releasing into natural habitats have been documented in fish and invertebrates, such as the Atlantic salmon *Salmo salar* (McDonald et al. 1998), European lobsters *Homarus gammarus* (van der Meeren 2001), blue crab *Callinectes sapidus* (Davis et al. 2005), and queen conch *Strombus gigas* (Stoner and Davis 1994). The learning capacity of Pacific threadfin *Polydactylus sexfilis* using reward conditioning showed that fish measuring 50 and 90 mm fork length (FL) learned better than the smaller (22 and 36 mm) or larger (130 mm) fish (Masuda and Ziemann 2000 as cited in Masuda 2004). This stage of high learning capacity corresponded with the stage of recruitment from offshore to coastal areas (Masuda 2004). In *Homarus gammarus*, normal development of claw morphology was stimulated by providing hard-shelled prey (Wickens 1986). In flounder *Paralichthys olivaceus*, conditioning was done by exposing them to small, non-predatory shore crabs *Matuta lunaris*, or larger ones behind a fence. Conditioned fish from either method were better able to avoid capture by crabs than naive fish, revealing that learning processes should play an important role in their predator avoidance (Hossain et al. 2002).

Habitats have also been examined prior to release not just for the suitability of the environment to the species for release but also for the presence of possible predators.

The importance of selecting suitable substrates has been stressed by Thomson et al. (1995) for scallop *Pecten fumatus*. Heavily silted or muddy substrate exposed to strong waves may cause scallops to disperse or die due to sedimentation or wave action. Ventilla (1982) further reported that *Patinopecten yessoensis* had the highest growth and survival on substrate with low mud content and at depths <40 m. Moreover, high mortality of scallops was attributed to the presence of starfish such as *Asterias amurensis* and *Distolasterias nipon*. Mortality was minimized by removing these starfish through dredging. Dance et al. (2003) demonstrated the effect of release habitat on survival of sandfish *Holothuria scabra* where higher recovery was observed from mangrove seagrass sites than from coral reef flats. Low survival in coral reef flats was accounted to predation by fish. Predation has been limited by putting released animals in cages or enclosures in the wild. Giant clams *Tridacna* spp. are protected by wire mesh cages when released in the wild until such time when they are no longer vulnerable to predators (Bell et al. 1997). *Trochus niloticus* were also reared in cages on a reef flat prior to release in the wild (Amos and Purcell 2003), while scallops *Pecten maximus* were placed in fenced areas to exclude crab predators (Strand et al. 2004). The use of cages as a temporary habitat for released animal is applicable only to molluscs since these organisms have very limited movement.

Genetic, disease and environmental considerations

The genetic effects of stock enhancement have been a controversial issue. The need for genetic resource management in stock enhancement programmes has been subject of intense public debate (Blankenship and Leber 1995). The major concern has been about the deleterious genetic effects of hatchery-reared animals on wild populations (Kitada and Kishino 2004). Since all releases have the potential to alter genetic structure of the target population, steps should be taken to minimize these changes (Bell et al. 2005). Some researchers have suggested strategies of minimizing genetic impacts on the wild population. According to McEachron et al. (1995), maintaining a large number of wild broodstock with yearly replenishment can minimize loss of genetic fitness due to inbreeding in the hatchery. In addition, Blaxter (2000) suggested that broodstock should be taken from the same habitat into which their offspring are to be released, in the expectation that natural selection will have ensured the survival of spawners most suited to that particular environment.

A responsible approach to stock enhancement requires that the negative impacts on the gene pools of wild populations be mitigated by the implementation of genetically sound breeding and release procedures (Tringali and Leber 1999). A number of population genetics methods are now available to help identify basic population structures. These include allozyme analysis, mitochondrial DNA analysis, polymerase chain reaction (PCR) and microsatellite analysis (Bell et al. 2005). The fate of released animals can now also be followed up using genetic markers which though labour-intensive and expensive are very useful and sensitive indicators in stock enhancement studies. Genetic tags developed from a rare mtDNA or allozyme alleles produced from broodstock with rare genotypes have been used in bay scallop *Argopecten irradians* (Seyoum et al. 2003) and abalone *Haliotis rufescens* (Gaffney et al. 1996). Increased frequency of such markers in the population after release of cultured juveniles indicates that restocking has made a contribution, whereas no change in frequency denotes failure.

Transfer of animals from one location to another or from the hatchery to the wild is a potential biosecurity risk thus, appropriate measures should be taken to destroy any possible pathogen or parasite these animals may carry with them. Hatchery diseases can be avoided or prevented through good husbandry. Diseases occurring in hatcheries can be lowered substantially through reduction of stress, provision of proper sanitation, high-quality water, natural food and a constant environment. The idea of getting broodstock from the same habitat where the juveniles will be released was suggested by Mushiake and Muroga (2004). These broodstock captured from the same habitat where the juveniles will be released may have survived infections prevailing in that area owing to their innate immunity. Despite any precaution, Sindermann (1993) recognized that it will never be possible to eliminate all disease risk associated with the propagation and translocation of marine invertebrates. According to Blankenship and Leber (1995), stock enhancement programmes that rely on intensive, large-scale hatchery production of juveniles have the potential to cause more harm than good unless great care is taken to minimize the inherent risks of promoting and transferring diseases. The potential for damage of transferred pathogens is not limited only to the wild conspecifics of released animals but also to probable atypical hosts (Sindermann 1993). Practical recommendations have been made for aquaculture, restocking and stock enhancement practices intended to reduce

disease risks to acceptable levels. These recommendations which encourage national management and regulatory authorities have been summarized by Bell et al. (2005) as follows: 1) promote the use of native species in aquaculture to reduce the need to rely on introduced species, thereby restricting the transfer networks for diseases; 2) develop and maintain maps of target host species showing the presence and abundance of each disease species as the basis for prohibition of transfers of broodstock and cultured animals from infected to non-infected area; 3) identify pathogen-free populations to be used as broodstock; 4) maintain proper sanitation, high-quality water and a constant environment in hatcheries to reduce stress; 5) test broodstock and progeny for known diseases in quarantine; 6) grant health certification to juveniles before release in the wild; 7) ensure that occurrence of any disease receives immediate attention which involve destruction of diseased animals and sterilization of facilities; and 8) use of smallest-sized animals that is practical in release programmes, because the risk of contracting diseases is lower for the smaller size classes in several species.

Aside from changes in genetic diversity of target stocks and the risk of introducing diseases, there are still other potential environmental impacts of stock enhancement. There is a great concern that changes in the relative abundance of the target species caused by successful large-scale releases may be at the expense of other valuable components of the ecosystem (Leber 1999). Removal of predators of the target species, like the dredging of starfish done in Japan (Ventilla 1982), may have undesirable effects on both plant and animal communities. Alteration of habitats made to favour target species may disadvantage other species. To favour intertidal aquaculture of Pacific oyster *Crassostrea gigas*, Olympia oyster *Ostrea lurida*, littleneck clam *Protothaca staminea* and Manila clam *Tapes* (=*Ruditapes*) *philippinarum*, the benthic communities in the Pacific Northwest estuaries have been subject to intense habitat restructuring. Habitat disturbances which strongly influence the carrying capacity for many fish and macroinvertebrates have been reported by Simenstad and Fresh (1995). These disturbances include adding gravel to mudflats and sandflats to enhance clam production, manipulating burrowing shrimp or eelgrass by manual harvesting or spraying oyster grounds with pesticides to kill populations of ghost shrimp *Neotrypaea* (=*Callianassa*) *californiensis* and mud shrimp *Upogebia pugettensis*, and manipulating oyster grounds like raking and levelling of intertidal

flats to improve oyster distribution on the plot. Introduction of new species into non-indigenous habitats has been widely practiced in Europe, Asia and Africa (Blaxter 2000). This intentional introduction or translocation of a species for the enhancement of stock has often unforeseen effects which may really be damaging to the recipient habitat. The introduction of the red king crab *Paralithodes camtschaticus* to the Barents Sea has the potential to substantially reduce the abundance of the native scallop *Chlamys islandica*, its prey species, after being able to establish a self-sustaining population that has expanded into Norwegian waters (Jorgensen 2005). All these impacts to the habitat and the communities within it should be taken into consideration before doing any stock enhancement activity.

To summarize, any stock enhancement activity changes the status quo of an ecosystem or the habitat involved. However, given the substantial damage these ecosystems have suffered due to anthropogenic activities and the depletion of fisheries resources in these ecosystems due to overfishing, the impact of adding juveniles aiming at improving production of the target species should not be a cause of great concern provided that this activity is conducted responsibly and that this will not cause further degradation to the ecosystem and its diversity.

Assessment

Assessing whether releases of cultured juveniles have achieved their objective is essential to responsible and adaptive management (Blankenship and Leber 1995; Leber 1999). Bell et al. (2005) proposed some indicators to estimate the contribution of cultured juveniles to the wild population which include: 1) the ratio of cultured juveniles to the estimated recruitment from the wild population; 2) the proportion of released animals in the commercial catch; 3) the survival rates of released animals at the size of harvest; and 4) the increases in total catch after enhancement. Marking animals prior to release is the most powerful way of estimating these contributions (Blankenship and Leber 1995).

Recent advances in tagging technology, with the introduction of an internally implanted, biologically inert, coded microwire tag (Jefferts et al. 1963), provide a basic tool for evaluating stock enhancement success (Blankenship and Leber 1995).

Tagging has become a necessity for stock enhancement to assess effectiveness of releases. It allows monitoring of survival, growth rate and even movement of individuals from recaptured released animals. Various tagging techniques have been used in different animals. The coded microwire tag (CWT) has been used in shrimps (NMT 2006), lobsters (Walker 1986), crabs (Ut 2002), and fish (Brennan et al. 2005). However, in molluscs, tagging may be external due to the presence of the shell. Topshell *Trochus niloticus* have been marked by gluing plastic tags to the shell (Castell et al. 1996). However, this tagging method underestimates survival of *Trochus* because of the highly cryptic nature of the juveniles. Hence, Crowe et al. (2001) developed a tagging system for *Trochus* employing the use of aluminum tags attached to the shells which can be monitored under water using a metal detector. Three-month old tropical abalone *Haliotis asinina* juveniles developed bluish-green coloured bands on their shells measuring 1.7, 2.6 and 4.2 mm wide after being fed artificial diets for 1, 2, and 3 weeks, respectively which can be used as good markers for stock enhancement (Gallardo et al. 2003). A more complex way of identifying abalone *Haliotis rubra* juveniles reared from parents of known genotypes is with the use of genetic markers (polymorphic microsatellite loci) (Conod et al. 2002). This technology allows assessment of the contribution of the progeny of hatchery-reared individuals to wild populations. Although labour-intensive and expensive, this is the most ideal tag for use in stock enhancement.

Status of stock enhancement initiatives

Successes and failures of stock enhancement programmes have been documented and reported in several species in different areas to serve as lessons, to ensure success for future endeavours and to avoid repeating past mistakes. However, some of these stock enhancement programmes, which due to lack of follow-up surveys or failure to recognize progenies, were not able to provide any evidence that released hatchery juveniles have contributed to the next generation (Bell et al. 2005).

One of the most successful stock enhancement programmes are in Japan. Ocean ranching of scallops increased landings from 5,000 tons around 1970 to 200,000 in the mid 1990s (Salvanes 2001). Contribution of released crabs in Japan varied from 9% in Okayama prefecture to 18% in Hamana Lake to as high as 59% in Osaka Bay

(Secor et al. 2002). Commercial landings of red sea bream in Kagoshima Bay from 1989 to 1991 were mainly from released fish, comprising 30% of catches in the central part of the bay to 64-83% in the inner part of the bay (Fushimi 2001). Kitada (1999) reported some 20-40% of the Japanese flounder *Paralichthys olivaceus* catches were from released fish. Red sea bream *Pagrus major* releases also appeared to maintain the catch in Kanagawa Prefecture and even enhance it in Kagoshima Prefecture (Blaxter 2000).

In Hawaii, cultured mullet *Mugil cephalus* released in 1990-1993 made significant contributions to the commercial striped mullet fishery in Kaneohe Bay (Leber and Lee 1997). The same was reported in one of the three released sites for the Pacific threadfin *Polydactylus sexfilis*, where released fish made a large contribution to juvenile abundance (Leber et al. 1998). In the Gulf of Mexico, the proportion of released red drum *Sciaenops ocellatus* in the total catches could be as high as 20% (Blaxter 2000).

Although Japan has been successful in stock enhancement of other species, attempts on abalone proved futile. According to Bell et al. (2005), despite moderately high post-releases survival rates of cultured juveniles in some areas of Japan, and good contributions of hatchery-reared abalone to annual landings, overall, abalone stock enhancement has failed to augment or even maintain levels of catches since releases began. The same happened in small-scale stock enhancement initiatives in California, where releases of abalone juveniles have not been effective (Burton and Tegner 2000). The programme for giant clam, on the other hand, has waned since the late 1980s and early 1990s, except for the Philippines and Japan, because of its being an expensive long-term exercise (Murakoshi 1986; Mingoa-Licuanan and Gomez 2002). Due to lack of appropriate sampling to detect juveniles, there is no firm evidence that restocked clams have contributed to the next generation. The lack of evidence of beneficial effects from releases had caused the closure of lobster hatcheries, which operated since almost a century ago in North America and 1955 in Europe (Bell et al. 2005). The difficulty in evaluating the contribution of released animals to the wild population is one of the most common problems encountered in stock enhancement.

Despite the increasing interest in the potential of marine stock enhancement in supplementing and replenishing depleted coastal fisheries, the debate continues over its usefulness in marine ecosystems (Leber et al. 1998). Careful evaluation must be done before making any enhancement on marine fish population which should be carried out responsibly (Blankenship and Leber 1995).

OBJECTIVES OF THE STUDY

General Objective

The main objective of the study was to conduct stock enhancement trials using three species of *Scylla* produced and reared in the Crustacean Hatchery of SEAFDEC Aquaculture Department, Tigbauan, Iloilo in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan where the mud crab population was reported to be overfished.

Specific Objectives:

From this general objective, specific objectives were drawn as follows:

1) To study the proposed mangrove habitat for stock enhancement; making a map of the area involved, showing the topography of the creeks, the heights above Chart Datum, the heights of the bank and the height of tidal water during the highest spring and neap tides. As assessment of the area of habitat suitable for mud crabs, mangrove communities throughout the site were mapped.

2) To study the population and fisheries of the wild population of the mud crabs *Scylla* spp. in the chosen study site; determining monthly mud crab production, identifying species present, sex ratios, monthly mean length and body weight in order to prove anecdotal claims of fishers that the wild mud crab population is declining.

3) To determine abundance and growth rates of the naturally-occurring wild crab population through a mark-recapture study using coded microwire tags which will serve as baseline data for comparing growth rates of hatchery-produced crabs when released, as the final experimental work.

4) To compare survival and growth rates of tagged mud crabs released both from the wild and hatchery-reared and study the effect of size-at-release and conditioning on survival of released crabs in order to establish protocols for future stock enhancement programmes.

CHAPTER 2

Mangrove community structure and topographical profile of the study site

INTRODUCTION

This chapter aims to describe in detail the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan which served as the study area from 2002 to 2005. It begins with a brief introduction about the Philippines, its mangroves, the uses and function of mangroves, and the causes of mangrove degradation. The results section describes the output of the mapping activity wherein the area suitable as mud crab habitat was determined. The mangrove species found in the area and the comparison of mangrove community structure between the three creeks are shown in tables and figures. The topographical profile of the creeks, environmental parameters measured and tidal inundation are described in detail.

The Philippines is an archipelago of about 7,100 islands. It is bordered by 17,460 km of coastline and 26.6 million ha of coastal waters (Primavera 2000). Marine resources are an important source of food and livelihood to more than half of the country's municipalities and villages which are situated along the coast. Panay Island, located in the centre of the Philippine Archipelago, is the sixth largest island in the country with a total land area of 12,394 km^2. On the northwest part of the island is the province of Aklan, one of the four provinces in Panay Island. The two most important mangrove areas in the province of Aklan are the reforested mangroves of New Buswang in the municipality of Kalibo (Walton et al. 2006a) and the natural mangroves of Naisud and Bugtong Bato in the municipality of Ibajay. Both mangroves cover an area of about 70 ha. However, the mangrove community in Kalibo has only 8 species, 4 replanted and 4 colonizers while the mangrove community in Ibajay has 27 species, all of them naturally occurring, except for some planting of *Nypa fruticans*.

Mangroves, a habitat or ecosystem where the land and the sea meet, is home not only to the so-called mangrove trees but also to a rich fauna derived from both the land and sea, most of them from the latter. Of the land animals, bats, birds and insects are the most conspicuous while among the marine organisms, crustaceans, molluscs and fish dominate (Macnae 1968; Saenger 2002). Mangrove ecosystems are reputed to be highly productive in spite of the relatively harsh environment in which they occur (Snedaker and Snedaker 1984). They are a valuable ecological and economic resource

(Naylor and Drew 1998). Traditionally, mangroves are a source of timber and fuel, food, medicine, toxicants and raw materials used in traditional fisheries (Bandaranayake 1998). They are important nursery grounds and breeding sites for birds, fish, crustaceans, shellfish, reptiles and mammals, renewable source of wood, accumulation sites for sediment, contaminants, carbon and nutrients, offer protection against coastal erosion, and are fast becoming tourist attractions (Mazda et al. 1997; Ewel et al. 1998; Alongi 2002; Sheridan and Hays 2003). Mangroves ameliorate the effects of storm surges, cyclones, and provide an interface between the freshwater wetland systems and the open sea (Bridgewater and Cresswell 1999). In a study conducted in Gazi Bay, Kenya, Fondo and Martens (1998) reported lower biodiversity in deforested areas than in natural mangroves, with other effects of deforestation including decreased availability of fish and prawns, increased coastal erosion and eventual reduction of seagrasses and coral reefs. The important role of mangroves as nurseries has been shown in the positive correlation between mangrove area and nearshore fisheries in the Philippines (Camacho and Bagarinao 1986), Malaysia (Macnae 1974) and Australia (Staples et al. 1985; Manson et al. 2005). In Mekong Delta in Vietnam, de Graaf and Xuan (1998) estimated that one ha of mangroves supports a marine catch of 450 kg yr^{-1} while Kathiresan and Rajendran (2002) in a study in India involving three mangrove areas, reported estimates of 11 kg ha^{-1} d^{-1} for shellfish and 4.5 kg ha^{-1} d^{-1} for finfish. The role of mangroves in coastal protection was demonstrated in the December 2004 tsunami that affected 13 Asian and African countries (Danielsen et al. 2005; Kar and Kar 2005; Kathiresan and Rajendran 2005). In Tamil Nadu, India, Kathiresan and Rajendran (2005) reported a significant negative correlation between the human death toll and the distance of human inhabitation from the sea, the elevation from mean sea level and the area of mangrove and other coastal vegetation. However, this findings was contested by Kerr et al. (2006). Re-analysing the values reported by Kathiresan and Rajendran (2005), Kerr et al. (2006) concluded that the apparent association of vegetation area on mortality was in fact due to a tendency for more vegetation to occur at higher elevations and, not surprisingly, to the greater potential areal extent of vegetation. Given a location of equal elevation and distance from the sea, differences in vegetation area did not mitigate human mortality caused by the tsunami. Moreover, Kerr et al. (2006) stressed that to expect the mangrove ecosystems to provide protection from large tsunamis appears unrealistic based on their re-analysis of Kathiresan and Rajendran (2005).

Nevertheless, the important functions of the mangrove ecosystem have often been unappreciated and not adequately valued economically (Naylor and Drew 1998; Ashton et al. 2003). Mangrove losses in the past two decades exceeded those for tropical rain forests and coral reefs (Valiela et al. 2005). Threats to mangrove existence include increasing population growth, global warming, aquaculture, and industrial and urban development (Bandaranayake 1998; Bridgewater and Cresswell 1999; Alongi 2002).

Increasing human population leads to encroachment in coastal areas which usually results in increased wastes dumped into the mangroves and adjacent coastal waterways. Natural processes such as global warming which cause sea level rise, and tsunamis which are capable of making drastic alterations to the seascape may also be detrimental to the mangroves once these changes expose the vegetation to continuous or prolonged inundation. Mangroves initial response to sea level rise is to migrate landward (Semeniuk 1994). However, this is only possible if the rate of sea level rise is slow. According to Rull et al. (1999), mangroves cannot sustain sea level rise with rates over 1.2 mm y^{-1} (12 cm in 100 years) and survival seemed to depend upon the balance between sedimentation from land and the rising sea level. Other short-term climatic events such as droughts and typhoons may also be important environmental forcing factors (Alongi 2002). These are some of the indirect causes of mangrove destruction. However, there are other activities that directly cause destruction of the habitat. Felling of forests is one of the oldest forms of commercial exploitation of mangroves. Although one good example of sustainable felling of mangroves is in the Matang Mangrove Forest Reserve in Perak, Malaysia, most felling is unsustainable (Alongi 2002), and also illegal. The loss of mangroves for pond aquaculture is currently one of the largest threats to mangrove forests worldwide, more so in the Philippines (Primavera 2000; Ronnback and Primavera 2000; Walters 2003). Some of the direct and indirect problems caused by pond aquaculture include immediate loss of mangroves for pond construction, blockage of tidal creeks, alteration of natural tidal flows, alteration of the groundwater table, increase in sedimentation rates and turbidity in natural waters, release of toxic wastes, overexploitation of wild seed stocks, development of acid sulphate soils, reduced water quality, introduction of excess nutrients, and alteration of natural food chains (Alongi 2002).

During the 1980s, mangroves were being protected or managed in many areas of the world, for the fisheries they support, the forest products they yield and the stability they contribute to the coastal zone, and yet, at the same time, they were being destroyed in other areas for reasons which are frequently illogical (Snedaker and Snedaker 1984). The latter was experienced in the Philippines. The Fisheries Decree of 1975 (Presidential Decree 704) mandated a policy to accelerate fishpond development, the Administrative Order 125 extended 10-year fishpond permits and leases to 25 years, and during the 'Shrimp Fever' of the 1980s, pond development further increased to 4,700 ha yr^{-1} (Primavera 2000). The Philippines lost 70% of its mangroves between 1918 and 1994 and was left with 120,500 ha of mangroves in 1994. Of these, only 2.5% belongs to Western Visayas, the region where Panay Island lies. The present study site is one of the few remaining natural mangroves on Panay Island, and is also at risk due to human activities.

Since our (SEAFDEC Mangrove Team) first sighting of the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, the area had always been considered as highly diverse and pristine. As viewed from the road, it is bordered by thickets of *Ceriops*, and mixed species of *Avicennia*, *Rhizophora*, *Xylocarpus* and *Bruguiera*. However, observations from regular visits to the site since 1995 revealed a slowly degrading ecosystem as evidenced by scattered burned and cut trees, some debarked tree trunks, patches of *Nypa fruticans*, and a slowly widening aquaculture pond.

Before considering the area for stock enhancement trials, it is necessary to assess the condition of the mangroves. According to Ashton et al. (2003), baseline ecological studies of mangroves are important for monitoring, management and conservation of the ecosystem. Moreover, stock enhancement is not suitable in a marine environment that is totally degraded (Blankenship and Leber 1995; Fushimi 1999). Hence, this study was conducted to have baseline information of the status of mangroves and the topographical profile of the creeks which will serve as habitat for mud crabs to be released for stock enhancement trials.

Specifically, the community structure of mangroves along the three creeks of Naisud River will be analysed in terms of relative density, relative frequency, relative dominance, importance value and species diversity index (English et al. 1997).

Mangrove distribution will be examined for possible zonation patterns, and major human activities that have caused mangrove destruction will be noted. The heights above Chart Datum of the creek beds and the heights of the bank where the quadrats for community structure study were laid out will be determined and their distances from the mouth of the river measured. Tidal inundation during high spring and high neap waters will also be established. Results of this study will not only be relevant to the present work being a proposed habitat for mud crab stock enhancement trials but may also be used to support efforts of local researchers, academics and non-governmental organizations (NGOs) in pushing for the declaration of this particular area as a mangrove forest reserve.

MATERIALS AND METHODS

The topographical profile of the area was surveyed to determine if the site is suitable for releasing mud crabs for stock enhancement trials. The study site was surveyed following steps suggested by Kjerfve (1990) for hydrological investigation of a mangrove wetland. These include, first, the determination of some geographical parameters such as the components of the whole landscape like the rivers, major constructions in the vicinity of the area, agricultural and other activities that have drainage in the system. The second step is the determination of the extent and area of the mangroves. And, third, is determining the topography of the area which includes measuring the vertical elevations within the system. The first two steps were conducted by mapping the study site (see below for details) and the results are presented in Figures 2.1 and 2.2. The third step was conducted on the three main creeks of Naisud River and along the banks where the quadrats were laid out and the results are presented in Figures 2.6, 2.7 and 2.8.

Study site

The mangrove study area is located along the three branches of Naisud River which separates the villages of Naisud and Bugtong Bato in Ibajay, Aklan, Philippines (122°11.4'-122°12.7'E longitude, 11°47.8'-11°48.5'N latitude). The first

Figure 2.1. A page from the GPS mapping software OziExplorer version 3.90.3a showing the area of the mangroves (70.2 ha) in Naisud and Bugtong Bato, Ibajay, Aklan, Philippines. The tracks were made around the edge of the mangrove forest and overlaid on an existing map of the area obtained from the National Mapping and Resource Information Authority (NAMRIA, Makati City, Philippines).

Figure 2.2. The most recent map of the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, output of the mapping and survey conducted in March and June-August 2005.

documentation of the mangrove species found in this area was in 1999, as part of an assessment study of two adjacent coastal municipalities of Tangalan and Ibajay, Aklan and was reported in Agbayani et al. (2001). Then, in 2002-2003, a more intensive work on species identification was conducted for the book on the mangroves of Panay Island (Primavera et al. 2004). The latter study reported 26 of the 35 species of mangroves found in the Philippines, 33 of which are found on Panay Island.

Mapping of the area

The area was first mapped in 1998 and has been extensively revised several times since then. The most extensive revisions were done in March 2005 (J.P. Altamirano and M.J.H.L. Lebata) and June-August 2005 (M.J.H.L. Lebata, K.J. Hutchinson and M.J. Langdown) and the map output has been used in the figures in the present work.

Mapping was done by first, walking around the periphery of the whole mangrove area to create tracks on Garmin GPS 12 XL. The output of this procedure was the outline and land area of the mangroves surveyed (Fig. 2.1). Tracks were then made in all creeks including small channels and passages branching from each creek. Important landmarks such as ponds, monospecific mangrove stands, *Nypa* plantation, settlement and cleared areas were marked. Using GPS mapping software, OziExplorer version 3.90.3a, these tracks and points were then overlaid on a map obtained from the National Mapping and Resource Information Authority (NAMRIA), Makati City, Philippines which shows the satellite image of the mangrove area in the villages of Naisud and Bugtong Bato. The map generated from the mapping software was enhanced by adding colours and textures using CorelDraw version 11.633 and Adobe Photoshop version 6. Within the mangrove area, areas occupied by ponds, *Nypa* plantation, monospecific stands were also demarcated. The final map (Fig. 2.2) has been modified depending on the need and used in the different chapters of the present study.

Mangrove community structure

Mangrove community structure was analysed following English et al. (1997). Data collection for mangrove quadrats was conducted in March-August 2005. Quadrats (10

x 10 m) were laid out at 100-m intervals from the first mangrove tree, nearest the mouth of Naisud River going upstream to the last mangrove tree on the boundary of the mangrove forest and the terrestrial vegetation. This was done in three of the four creeks of Naisud River, on the left and right sides of the creeks facing downstream whenever possible (Fig. 2.3). Each quadrat was identified with numbers starting from one at the mouth of the river or the junction where the two creeks (A and B) branched out from the main creek (Creek C). During sampling, additional quadrats (J1, J2 and J3) were inserted in between the numbered quadrats when a distinct change in species composition between the two succeeding quadrats was observed and labelled as such in the maps, graphs and tables. The middle creek which is the source of freshwater coming from the mountains was excluded because of its sandy-rocky nature and mangroves grow only for a short distance from the junction where all the three creeks converge. Also, no crab fishing activity is being done in this area because of the nature of its substrate.

The quadrats were laid out in such a way that the origin was always at the upper left hand corner when facing the creek. Four pre-measured strings (11 m each marked 0.5 m from both ends for tying to the corner of the quadrat) were used in making the quadrats. From the origin, two strings were tied, one pulled straight to the back and the other to the right making 2 sides of the quadrat. From each of the corners where the two strings ended, the remaining two strings were tied to meet at the lower right corner of the quadrat forming a 10 x 10 m area. Inside the 10 x 10 m quadrat, two smaller subquadrats (5 x 5 m and 2 x 2 m) were made starting from the origin. Within the 100 m^2 quadrat, all trees were sampled, while the saplings and seedlings were sampled inside the 25 m^2 and the 4 m^2 quadrats, respectively (Fig. 2.3). Seedlings are plants <1 m in height, saplings are plants >1 m in height with girth at breast height (GBH)<4 cm, and trees are plants >1 m in height with GBH>4 cm (English et al. 1997). Girth at breast height of every tree and all the branches with GBH>4 cm were measured and the species identified. Breast height is constant at 1.3 m from the ground. Saplings and seedlings were also identified and counted within the designated subquadrats.

Figure 2.3. Map of the mangroves in Naisud and Bugtong Bato, Ibajay, Aklan, Philippines showing the location of the quadrats where community structure analysis was done, the location of the tidal markers where the high water neap and spring levels were measured, and an enlarged sample of a quadrat showing its measurements, the plants measured within each subquadrat and its origin with reference to the creek.

Each quadrat was analysed for relative density, frequency and dominance, importance value and species diversity using the equations from Cintron and Novelli (1984) and English et al. (1997) as follows:

1) Relative density

$$RDen = \frac{Sp_n}{Sp_T} \times 100\,; \qquad \text{(Equation 1)}$$

where: $RDen$ = relative density

Sp_n = number of a particular species in a quadrat

Sp_T = total number of all species in a quadrat

2) Relative frequency

$$RFre = \frac{Sp_f}{\sum Sp_T} \times 100\,; \qquad \text{(Equation 2)}$$

where: $RFre$ = relative frequency

Sp_f = frequency of a species in a quadrat

$\sum Sp_T$ = sum of frequency of all species in a quadrat

And the frequency of a species is obtained as follows:

$$Sp_f = \frac{P_{Sp}}{P_T} \times 100\,; \qquad \text{(Equation 3)}$$

where: Sp_f = frequency of a species

P_{Sp} = number of quadrats occupied by a species

P_T = total number of quadrats sampled, in this case per creek

3) Relative dominance

$$RDom = \frac{Sp_{BA}}{Sp_{TBA}} \times 100; \qquad \text{(Equation 4)}$$

where: $RDom$ = relative dominance

Sp_{BA} = basal area of a species in a quadrat

Sp_{TBA} = total basal area of all species in a quadrat

Basal area expressed in cm^2 is obtained as follows:

$$BA = \frac{\pi DBH^2}{4}; \qquad \text{(Equation 5)}$$

where: BA = basal area, calculated for each trunk or branch of each species

π = pi; 3.1416

DBH = diameter at breast height

And the diameter at breast height in cm was obtained by:

$$DBH = \frac{GBH}{\pi}; \qquad \text{(Equation 6)}$$

where: GBH = girth at breast height, measurement taken from each tree during sampling

4) Importance value

$$IV = RDen + RFre + RDom; \qquad \text{(Equation 7)}$$

where: IV = importance value

$RDen$ = relative density

$RFre$ = relative frequency

$RDom$ = relative dominance

5) Species diversity

$$H = -\sum_{Sp=1}^{T}\left(\frac{IV_{Sp}}{IV_T}\right)\log\left(\frac{IV_{Sp}}{IV_T}\right); \qquad \text{(Equation 8)}$$

where: T = total number of species in a quadrat

IV_{Sp} = importance value of species Sp in a quadrat

IV_T = sum of importance values of all species in a quadrat

Topographical profile

The topographical profile of the creeks that provide the mangroves access to the sea was surveyed. The heights of the three creek beds from the Chart Datum and the heights of the banks where the quadrats were laid out were measured using Leica NA 820 geosystems automatic level mounted on a tripod. Creek heights were obtained from the centre of the creek using a metered staff to determine changes in topography which was viewed through the reticule of the automatic level. The measurement started at the mouth of Naisud River on a low tide to determine the height of the starting point based on the values from the tide table and was marked as height above Chart Datum. The heights of the banks were measured on both sides wherever possible. In some areas, the measurement was done only on one side due to the presence of lobster mounds or ponds on the other side which made it difficult to find the level of the actual bank.

Environmental parameters

Water temperature and salinity in the area were measured using a hand-held YSI conductivity meter during the monthly sampling of crabs and reported in subsequent chapters. Readings were taken on each creek during high tide in the day. Pore water salinity, soil pH and sediment particle size were measured in all quadrats during the community structure analysis and reported by Hutchinson (2005).

Tidal inundation

Tidal inundation heights for both high water spring and high water neap were determined by Langdown (2005). From the topographical profile done earlier, the highest and lowest points on each creek were selected for determining tidal inundation heights following the procedure described by English et al. (1997). Both measurements were taken during the highest spring and highest neap tides of the year which were in July 2005. The highest spring tide of the year when the measurements were taken was on 23 July 2005 at 1209 hr with the highest tide predicted from the tide table of 2.3 m. During the highest neap tide, the water did not reach the banks of the creek where the stations for water mark levels were posted so the high water neap

(HWN) was obtained from the predicted height in the tide table and was adjusted based on the error obtained during the high water spring (HWS) measurements.

Data analysis

Data obtained from the quadrats for the community structure analysis and those from the topographical profile were analysed and plotted using Minitab version 14. The use of left and right on the quadrats and on the creeks always mean facing the mouth of Naisud River as the reference point.

RESULTS

Mapping

A total area of 70.2 ha of mangroves in Naisud and Bugtong Bato, Ibajay, Aklan, Philippines was estimated from the area within the track obtained around the edge of the forest using GPS mapping software OziExplorer version 3.90.3a (Fig. 2.1). The middle creek of Naisud River which stretches from its mouth going upstream to the mountains divides the mangroves to the villages of Naisud where approximately 35% belongs and the remaining area to Bugtong Bato. The map drawn from this activity is shown in Figure 2.2 as a form of biotope map. Areas observed to be dominated by a single species or mixed species from the same genus were illustrated on the map. The map shows the expanding plantation of *Nypa fruticans* which occupies about 30% of the area, the climax forest of large, old *Avicennia rumphiana/Avicennia officinalis* surrounded by a mixed species zone, the area occupied by thick thickets of *Ceriops decandra*, the mixed *Avicennia* spp. area, the *Avicennia marina* zone, the *Bruguiera sexangula* area, the *Ceriops tagal* plantation, and the ponds, settlement and cleared areas. Excluding the area occupied by the *Nypa* plantation, settlement and cleared areas which cover approximately 20 ha, the remaining area left occupied by the natural mangroves is roughly 50 ha. The large pond measuring 15 ha on the west side of the study site, running parallel along Creek A, may have been part of the mangrove forest in the past.

The map further showed the creeks where most of the collection and releases of crabs were conducted. The creek on the west named Creek A is the branch of Naisud River that supplies water to the mangroves in Naisud as well as the culture pond running parallel along its left side. The creek on the east named Creek C supplies water to the mangroves of Bugtong Bato and the smaller ponds in its upper reaches. Creek B is a branch of Creek C (Fig. 2.3).

During the survey and mapping activity, it was learned from the villagers that most of the ponds are owned by government officials either occupying government seats in the past or presently active. Productive ponds were observed to be separated from the creek by a narrow dike then by a thin strip (approximately 5 m) of mangroves. Abandoned ponds were found to be colonized by *Ceriops decandra* seedlings. The *Nypa fruticans* plantations were claimed by the "owners" as theirs and that the land had been handed down over many generations. Although people in the coastal areas are now aware that mangrove cutting is prohibited, scattered cases of cut trees were still observed. To silently cut the trees, smaller trees are cut using a machete while larger trees are killed first before being cut by burning the trunk or removing the bark. Once it is dead, they wait for it to fall before chopping it. The creeks also serve as a sewage disposal area for all the wastes coming from the ponds, the settlement area and the newly built piggery, some 100 m away from Creek A.

Mangrove community structure

A total of 48 quadrats were laid out along the three creeks, 43 were originally marked for sampling during the March 2005 mapping and 5 were added in between some quadrats where a distinct change in species composition was observed. In Creek A, 16 quadrats have been sampled, 7 on the left and 9 on the right. In Creek B, 7 quadrats have been sampled, 4 on the left and 3 on the right. In the main creek, Creek C, 25 quadrats have been sampled, 14 on the left and 11 on the right (Fig. 2.3).

Of the 27 species of mangroves inhabiting the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, 19 species belonging to 12 genera were sampled in the 48 quadrats laid out. These were *Aegiceras corniculatum*, *Avicennia alba*, *Avicennia marina*, *Avicennia officinalis*, *Avicennia rumphiana*, *Bruguiera cylindrica*, *Bruguiera*

gymnorrhiza, *Bruguiera parviflora*, *Camptostemon philippinensis*, *Ceriops decandra*, *Excoecaria agallocha*, *Heritiera littoralis*, *Nypa fruticans*, *Rhizophora apiculata*, *Rhizophora mucronata*, *Scyphiphora hydrophyllacea*, *Sonneratia alba*, *Xylocarpus granatum* and *Xylocarpus moluccensis* (Table 2.1). Most of these species are trees, except for *Aegiceras corniculatum*, *Ceriops decandra* and *Scyphiphora hydrophyllacea* which are shrubs and *Nypa fruticans* which is a palm. The code of each species sampled from the quadrats and used in Fig. 2.5 was adapted from English et al. (1997).

Three species of *Acanthus* which were not included in the sampling and analysis of species composition were counted and included in the list of mangrove species found in the area (Table 2.1). These are small creeping plants, which are characteristic ground flora of the mangroves. Although these shrubs are considered only as mangrove associates by Tomlinson (1994), they were listed as true mangrove species by Primavera et al. (2004) because of their complete fidelity to the mangrove area, a characteristic possessed by a true mangrove. They were widespread in the study site, especially in areas with open access to sunlight and were observed in some of the quadrats sampled.

To minimize repetition in explaining the results, the tables for relative density, dominance and diversity and importance value will be described here before going into the details for each creek. For tables showing no value for relative dominance, this means that no tree was recorded in that quadrat, since dominance values are obtained from basal areas which are calculated in turn from the diameter of the tree trunks. This is commonly observed in the additional quadrats inserted (J1, J2, J3) between the numbered quadrats because they were dominated by *Nypa fruticans* or in some quadrats dominated mostly by seedlings, saplings or mangroves that are shrubby in nature. Therefore, it follows that quadrats with no relative dominance values will result in a 200% total importance value since the importance value is the sum of relative density (100%), frequency (100%) and dominance (100%) (see Equation 7). Quadrats with relative dominance values have 300% total importance values. The tables for these indices are preceded by an enlarged section of the map showing a particular creek sampled to serve as reference for checking the location of the quadrats on the banks of the creek.

Table 2.1. Species of true mangroves in the Philippines (Tomlinson 1994)[a], in Naisud and Bugtong Bato, Ibajay, Aklan (Primavera et al. 2004)[b], and those observed during mapping and in the quadrats laid out in Creeks A, B, and C. Note the first sighting of *Bruguiera parviflora* in the study area. Mangrove species in boldface are not found in Panay Island.

Species	Philippine Species[a]	Naisud & Bugtong Bato Species[b]	Present Study			
			Mapping	A	B	C
1. *Acanthus ebracteatus*	✓	✓	✓	✓	✓	✓
2. *A. ilicifolius*	✓	✓	✓	✓	✓	✓
3. *A. volubilis*	✓	✓	✓	✓	✓	✓
4. ***Aegialitis annulata***	✓	×	×	×	×	×
5. *Aegiceras corniculatum*	✓	✓	✓	✓	×	✓
6. *A. floridum*	✓	×	×	×	×	×
7. *Avicennia alba*	✓	✓	✓	✓	×	✓
8. *A. marina*	✓	✓	✓	✓	✓	✓
9. *A. officinalis*	✓	✓	✓	✓	✓	✓
10. *A. rumphiana*	✓	✓	✓	✓	✓	✓
11. *Bruguiera cylindrica*	✓	✓	✓	✓	✓	✓
12. *B. gymnorrhiza*	✓	✓	✓	×	✓	✓
13. *B. parviflora*	✓	×	✓	×	×	✓
14. *B. sexangula*	✓	✓	✓	×	×	×
15. *Camptostemon philippinensis*	✓	✓	✓	✓	✓	✓
16. *Ceriops decandra*	✓	✓	✓	✓	✓	✓
17. *C. tagal*	✓	✓	✓	×	×	×
18. *Excoecaria agallocha*	✓	✓	✓	✓	×	✓
19. *Heritiera littoralis*	✓	✓	✓	×	✓	✓
20. ***Kandelia candel***	✓	×	×	×	×	×
21. *Lumnitzera littorea*	✓	✓	✓	×	×	×
22. *L. racemosa*	✓	✓	✓	×	×	×
23. *Nypa fruticans*	✓	✓	✓	✓	✓	✓
24. *Osbornia octodonta*	✓	×	×	×	×	×
25. *Pemphis acidula*	✓	×	×	×	×	×
26. *Rhizophora apiculata*	✓	✓	✓	✓	×	✓
27. *R.* x *lamarckii*	✓	×	×	×	×	×
28. *R. mucronata*	✓	✓	✓	✓	×	✓
29. *R. stylosa*	✓	✓	✓	×	×	×
30. *Scyphiphora hydrophyllacea*	✓	✓	✓	✓	×	×
31. *Sonneratia alba*	✓	✓	✓	✓	×	✓
32. *S. caseolaris*	✓	×	×	×	×	×
33. *S. ovata*	✓	×	×	×	×	×
34. *Xylocarpus granatum*	✓	✓	✓	✓	✓	✓
35. *X. moluccensis*	✓	✓	✓	✓	✓	✓
TOTAL	**35**	**26**	**27**	**19**	**14**	**21**

Creek A – Sixteen species of mangroves were sampled in the quadrats in Creek A namely *Aegiceras corniculatum*, *Avicennia alba*, *Avicennia marina*, *Avicennia officinalis*, *Avicennia rumphiana*, *Bruguiera cylindrica*, *Camptostemon philippinensis*, *Ceriops decandra*, *Excoecaria agallocha*, *Nypa fruticans*, *Rhizophora apiculata*, *Rhizophora mucronata*, *Scyphiphora hydrophyllacea*, *Sonneratia alba*, *Xylocarpus granatum* and *Xylocarpus moluccensis* (Table 2.1). Of these 16 species, *Avicennia alba*, *Excoecaria agallocha*, *Rhizophora apiculata* and *Scyphiphora hydrophyllacea* were observed only on the left side of the creek while *Aegiceras corniculatum*, *Bruguiera cylindrica* and *Rhizophora mucronata* on the right.

Nypa fruticans was the most dense species in nine of the 16 quadrats, followed by *Ceriops decandra* which was most dense in six quadrats and *Avicennia officinalis* in only one quadrat (Table 2.2a). *Avicennia marina* was present at relatively higher density compared with the other *Avicennia* spp. in the first two quadrats. However, in quadrats 3-8 *Avicennia officinalis* and *Avicennia rumphiana* were denser. Although relative frequency is more evenly distributed among species compared with relative density (Table 2.2b), *Avicennia rumphiana* and *Ceriops decandra* were the most frequent species in the first four quadrats and alternately dominate most of the quadrats towards the end of Creek A, except for quadrats J1 and J3 where *Nypa fruticans* and *Avicennia officinalis* were the most frequent, respectively. *Avicennia marina* was the most dominant species in quadrat 1, *Avicennia rumphiana* in quadrats 2-5 and 8, while *Avicennia officinalis* and *Sonneratia alba* dominated the other four quadrats (Table 2.2c). Other species such as *Camptostemon philippinensis* and the two species of *Xylocarpus* had remarkable contributions in some of the quadrats they occupied while some species like *Aegiceras corniculatum*, *Avicennia alba*, *Bruguiera cylindrica*, *Excoecaria agallocha*, *Rhizophora apiculata*, *Rhizophora mucronata* and *Scyphiphora hydrophyllacea* contributed less since they occupy only one or two quadrats in the whole of Creek A.

The importance value of each species in their respective quadrats is shown in Table 2.2d. In the first two quadrats on the left *Ceriops decandra* was the most important species, while *Avicennia marina* was the most important species in the first quadrat on the right, and from third quadrat on the right down to quadrat 5, *Avicennia rumphiana*

Reference for quadrats 1 – 8, J1, J2 and J3 in Creek A

Table 2.2 a) Relative density, b) relative frequency, c) relative dominance, and d) importance value of the mangrove species sampled in the quadrats on the left (L) and right (R) side of Creek A in the mangroves of Naisud, Ibajay, Aklan, Philippines. Values in boldface are the highest in their respective quadrats.

a) Relative density (quadrats 1 to 4)

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L	4-R
A. corniculatum	0.00	4.27	0.00	0.00	0.00	1.22	0.00	0.00
A. alba	2.33	0.00	0.00	0.00	1.64	0.00	0.00	0.00
A. marina	11.63	24.79	0.00	15.94	1.64	9.76	0.00	7.87
A. officinalis	9.30	0.85	0.58	0.00	0.00	1.22	0.00	0.79
A. rumphiana	1.16	6.84	1.74	14.49	31.15	34.15	**55.00**	2.36
B. cylindrica	0.00	0.85	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	1.16	2.56	0.58	1.45	0.00	1.22	30.00	0.00
C. decandra	**66.28**	23.93	**95.35**	**66.67**	**60.66**	**46.34**	5.00	5.51
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	5.00	0.00
N. fruticans	1.16	**32.48**	0.00	0.00	0.00	0.00	0.00	**80.31**
R. apiculata	1.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	1.22	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	5.00	0.00
S. alba	4.65	2.56	0.00	1.45	1.64	1.22	0.00	0.00
X. granatum	0.00	0.00	1.74	0.00	3.28	0.00	0.00	0.00
X. moluccensis	1.16	0.85	0.00	0.00	0.00	3.66	0.00	3.15

a) Relative density (quadrats 5 to 8)

Species/Quadrat	5-R	J1-L	6-L	J2-R	7-L	7-R	J3-R	8-R
A. corniculatum	0.53	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	13.23	0.00	2.50	11.67	0.00	5.71	17.42	24.62
A. rumphiana	24.34	0.00	2.50	0.00	6.32	0.00	0.00	7.69
B. cylindrica	0.00	0.00	0.00	1.67	0.00	0.00	0.00	0.00
C. philippinensis	0.00	0.00	0.00	0.00	0.00	31.43	0.00	0.00
C. decandra	0.00	0.00	0.00	3.33	3.45	**45.71**	0.00	3.08
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	**59.26**	**100.00**	**92.50**	**78.33**	**89.66**	0.00	**81.82**	**63.08**
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	2.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	2.50	3.33	0.57	0.00	0.00	0.00
X. granatum	0.00	0.00	0.00	1.67	0.00	0.00	0.00	1.54
X. moluccensis	0.53	0.00	0.00	0.00	0.00	17.14	0.76	0.00

b) Relative frequency (quadrats 1 to 4)

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L	4-R
A. corniculatum	0.00	3.85	0.00	0.00	0.00	4.41	0.00	0.00
A. alba	2.63	0.00	0.00	0.00	4.55	0.00	0.00	0.00
A. marina	7.89	7.69	0.00	13.33	13.64	8.82	0.00	10.34
A. officinalis	14.47	14.10	23.91	0.00	0.00	16.18	0.00	18.97
A. rumphiana	**15.79**	**15.38**	**26.09**	**26.67**	**27.27**	**17.65**	**36.36**	**20.69**
B. cylindrica	0.00	2.56	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	9.21	8.97	15.22	15.56	0.00	10.29	21.21	0.00
C. decandra	**15.79**	**15.38**	**26.09**	**26.67**	**27.27**	**17.65**	**36.36**	**20.69**
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	3.03	0.00
N. fruticans	13.16	12.82	0.00	0.00	0.00	0.00	0.00	17.24
R. apiculata	1.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	2.94	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	3.03	0.00
S. alba	10.53	10.26	0.00	17.78	18.18	11.76	0.00	0.00
X. granatum	0.00	0.00	8.70	0.00	9.09	0.00	0.00	0.00
X. moluccensis	9.21	8.97	0.00	0.00	0.00	10.29	0.00	12.07

b) Relative frequency (quadrats 5 to 8)

Species/Quadrat	5-R	J1-L	6-L	J2-R	7-L	7-R	J3-R	8-R
A. corniculatum	6.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	24.44	0.00	26.83	23.40	0.00	29.73	**39.29**	22.45
A. rumphiana	**26.67**	0.00	**29.27**	0.00	**28.57**	0.00	0.00	**24.49**
B. cylindrica	0.00	0.00	0.00	4.26	0.00	0.00	0.00	0.00
C. philippinensis	0.00	0.00	0.00	0.00	0.00	18.92	0.00	0.00
C. decandra	0.00	0.00	0.00	**25.53**	**28.57**	**32.43**	0.00	**24.49**
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	22.22	**100.00**	24.39	21.28	23.81	0.00	35.71	20.41
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	4.44	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	19.51	17.02	19.05	0.00	0.00	0.00
X. granatum	0.00	0.00	0.00	8.51	0.00	0.00	0.00	8.16
X. moluccensis	15.56	0.00	0.00	0.00	0.00	18.92	25.00	0.00

c) Relative dominance (quadrats 1 to 4)

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L	4-R
A. corniculatum	0.00	3.71	0.00	0.00	0.00	2.71	0.00	0.00
A. alba	20.60	0.00	0.00	0.00	0.41	0.00	0.00	0.00
A. marina	**27.81**	**36.75**	0.00	16.80	0.70	17.49	0.00	12.65
A. officinalis	19.12	13.02	48.13	0.00	0.00	4.41	0.00	2.60
A. rumphiana	3.90	14.99	**48.77**	**80.84**	**96.57**	**67.11**	**98.68**	**82.38**
B. cylindrica	0.00	5.40	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	3.86	1.16	3.09	1.10	0.00	1.49	0.23	0.00
C. decandra	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	2.66	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	1.10	0.00
S. alba	20.51	24.97	0.00	1.26	0.47	4.14	0.00	0.00
X. granatum	0.00	0.00	0.00	0.00	1.85	0.00	0.00	0.00
X. moluccensis	4.21	0.00	0.00	0.00	0.00	0.00	0.00	2.36

c) Relative dominance (quadrats 5 to 8)

Species/Quadrat	5-R	J1-L	6-L	J2-R	7-L	7-R	J3-R	8-R
A. corniculatum	13.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	0.00	0.00	**65.33**	13.67	0.00	**83.51**	0.00	0.75
A. rumphiana	**58.29**	0.00	29.95	0.00	0.00	0.00	0.00	**97.63**
B. cylindrica	0.00	0.00	0.00	5.38	0.00	0.00	0.00	0.00
C. philippinensis	0.00	0.00	0.00	0.00	0.00	15.61	0.00	0.00
C. decandra	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.84
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	28.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	4.72	**72.91**	**100.00**	0.00	0.00	0.00
X. granatum	0.00	0.00	0.00	8.04	0.00	0.00	0.00	0.77
X. moluccensis	0.00	0.00	0.00	0.00	0.00	0.87	0.00	0.00

d) Importance value (quadrats 1 to 4)

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L	4-R
A. corniculatum	0.00	11.83	0.00	0.00	0.00	8.34	0.00	0.00
A. alba	25.56	0.00	0.00	0.00	6.59	0.00	0.00	0.00
A. marina	47.33	**69.23**	0.00	46.07	15.97	36.07	0.00	30.87
A. officinalis	42.90	27.97	72.63	0.00	0.00	21.81	0.00	22.36
A. rumphiana	20.85	37.21	76.60	**122.00**	**154.99**	**118.90**	**190.04**	**105.43**
B. cylindrica	0.00	8.82	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	14.23	12.70	18.89	18.11	0.00	13.00	51.44	0.00
C. decandra	**82.07**	39.32	**121.44**	93.33	87.93	63.99	41.36	26.20
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	8.03	0.00
N. fruticans	14.32	45.30	0.00	0.00	0.00	0.00	0.00	97.56
R. apiculata	2.48	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	6.82	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	9.13	0.00
S. alba	35.68	37.79	0.00	20.48	20.29	17.12	0.00	0.00
X. granatum	0.00	0.00	10.44	0.00	14.22	0.00	0.00	0.00
X. moluccensis	14.58	9.83	0.00	0.00	0.00	13.95	0.00	17.58
No. of Species	**10**	**10**	**5**	**5**	**6**	**9**	**5**	**6**

d) Importance value (quadrats 5 to 8)

Species/Quadrat	5-R	J1-L	6-L	J2-R	7-L	7-R	J3-R	8-R
A. corniculatum	20.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	37.67	0.00	94.65	48.74	0.00	**118.96**	56.71	47.82
A. rumphiana	**109.29**	0.00	61.72	0.00	34.89	0.00	0.00	**129.82**
B. cylindrica	0.00	0.00	0.00	11.30	0.00	0.00	0.00	0.00
C. philippinensis	0.00	0.00	0.00	0.00	0.00	65.96	0.00	0.00
C. decandra	0.00	0.00	0.00	28.87	32.02	78.15	0.00	28.41
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	81.48	**200.00**	**116.89**	**99.61**	113.46	0.00	**117.53**	83.49
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	35.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. hydrophyllacea	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	26.73	93.26	**119.62**	0.00	0.00	0.00
X. granatum	0.00	0.00	0.00	18.22	0.00	0.00	0.00	10.48
X. moluccensis	16.08	0.00	0.00	0.00	0.00	36.93	25.76	0.00
No. of Species	**6**	**1**	**4**	**6**	**4**	**4**	**3**	**5**

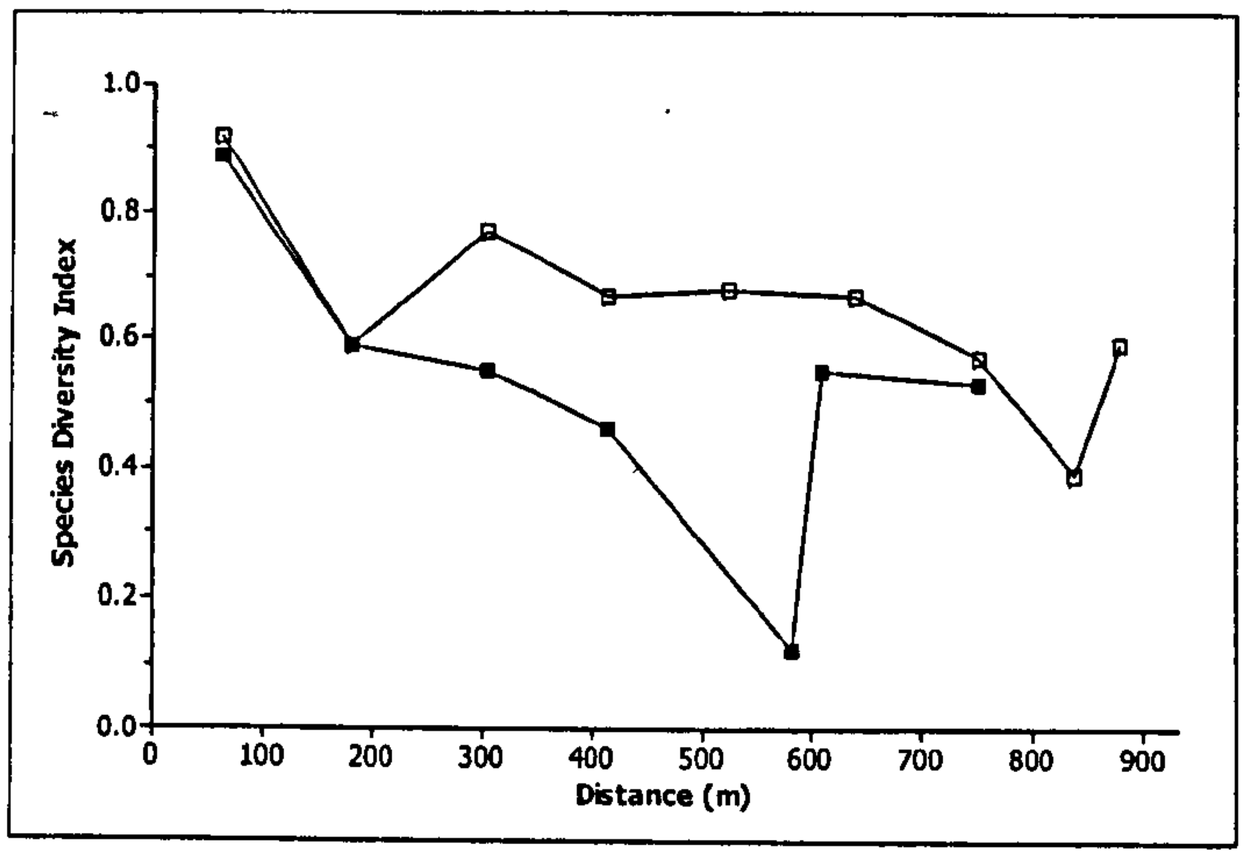
was the most important. *Avicennia marina* and *Ceriops decandra* also contributed a significant share of importance next to *Avicennia rumphiana* in these quadrats. All the additional quadrats (J1, J2 and J3) were dominated by *Nypa fruticans* while quadrat 7 had *Sonneratia alba* on the left and *Avicennia officinalis* on the right as the most important species. In general, the *Avicennia* spp. were the most important species in Creek A with lesser but relevant contributions from *Xylocarpus* spp., *Camptostemon philippinensis* and *Sonneratia alba* in some of the quadrats.

The highest species diversity values obtained for Creek A (Fig. 2.4a) were in quadrat 1, 0.89 in the left and 0.92 in the right. A decreasing trend was observed from quadrat 1 to quadrat 8 with the lowest diversity of 0.12 recorded in quadrat J1 which is dominated by *Nypa fruticans*. Species diversity on the right side of the creek was negatively correlated with distance from the river mouth (Pearson correlation=-0.74, $p<0.05$) and the height of the river bed above Chart Datum (Pearson correlation=-0.68, $p<0.05$). There was no correlation between distance and height of river bed with species diversity on the left side of the creek.

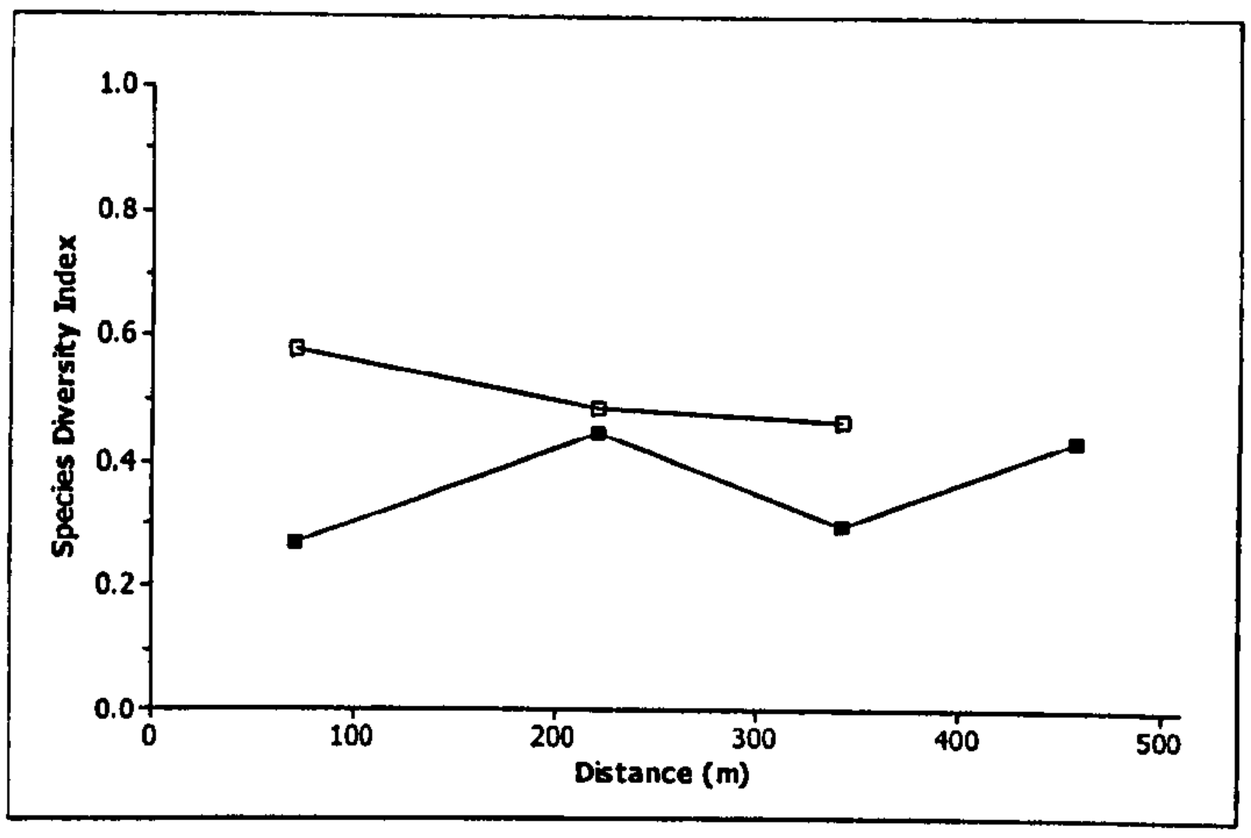
Creek B – Eleven species of mangroves were sampled in the quadrats in Creek B namely *Avicennia marina*, *Avicennia officinalis*, *Avicennia rumphiana*, *Bruguiera cylindrica*, *Bruguiera gymnorrhiza*, *Camptostemon philippinensis*, *Ceriops decandra*, *Heritiera littoralis*, *Nypa fruticans*, *Xylocarpus granatum* and *Xylocarpus moluccensis* (Table 2.1). Of these 11 species, *Bruguiera cylindrica* and *Heritiera littoralis* were observed only on the left side of the creek while *Avicennia marina*, *Bruguiera gymnorrhiza*, *Camptostemon philippinensis*, *Ceriops decandra* and *Xylocarpus granatum* on the right.

On the left side of the creek, *Nypa fruticans* had the highest relative density in three out of four quadrats (1, 3 and 4) while *Avicennia officinalis* was the densest species in quadrat 2. The right side of the creek was more diverse, with different species having the highest relative density in different quadrats; *Avicennia officinalis* in quadrat 1, *Ceriops decandra* in quadrat 2 and *Avicennia rumphiana* in quadrat 3 (Table 2.3a). In terms of relative frequency, although the differences between each species are less compared with relative density, *Nypa fruticans* was the most frequent species in six out of seven quadrats in Creek B, except for quadrat 2 on the right where *Avicennia*

a) Creek A



b) Creek B



c) Creek C

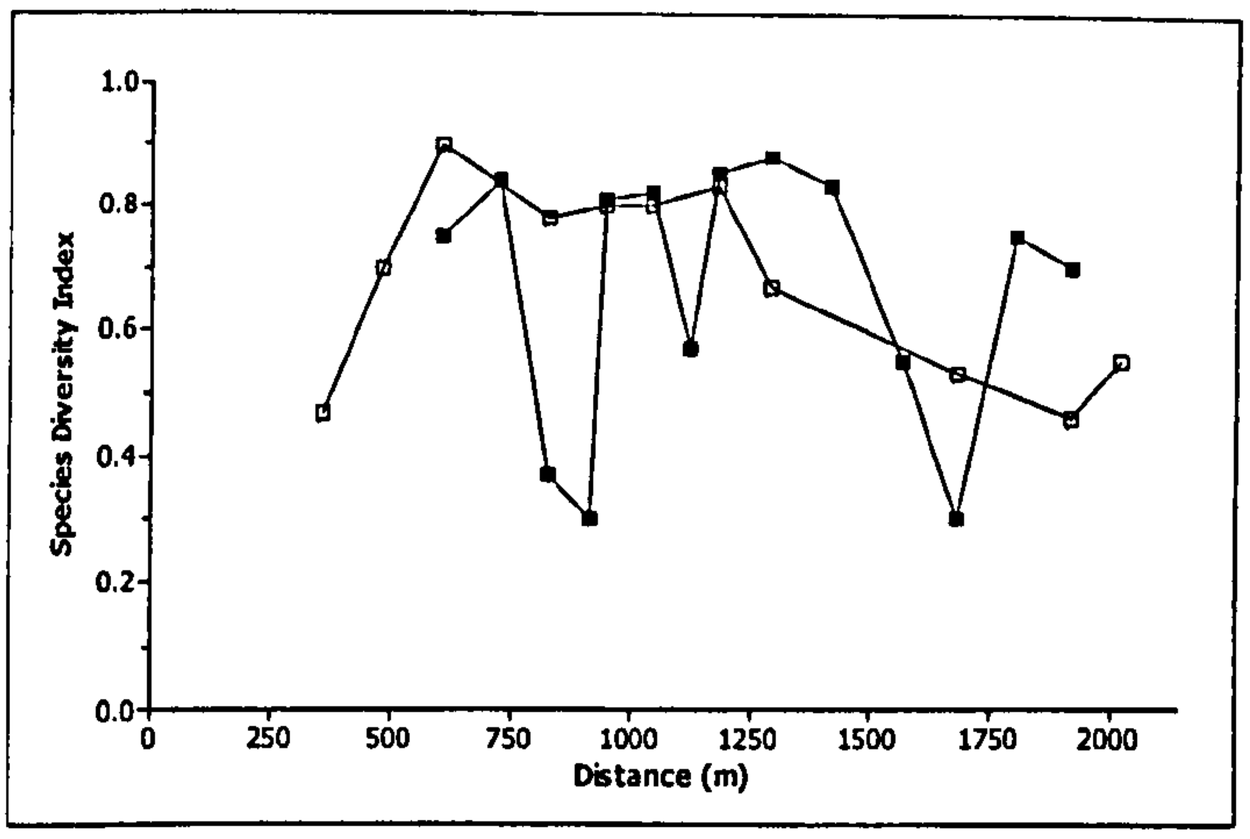


Figure 2.4. Species diversity of the quadrats sampled in Creeks A, B and C of the mangroves in Naisud and Bugtong Bato, Ibajay, Aklan, Philippines. Filled squares represent quadrats to the left side of the creek and empty squares quadrats to the right.

Reference for quadrats 1 – 4 in Creek B

Table 2.3. a) Relative density, b) relative frequency, c) relative dominance, and d) importance value of mangrove species sampled in the quadrats on the left (L) and right (R) sides of Creek B in the mangroves of Bugtong Bato, Ibajay, Aklan, Philippines. Values in boldface are the highest in their respective quadrats.

a) Relative density

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L
A. marina	0.00	2.08	0.00	0.00	0.00	0.00	0.00
A. officinalis	2.62	**52.08**	**56.00**	0.00	0.00	0.00	1.41
A. rumphiana	0.00	0.00	0.00	34.78	5.88	**61.54**	0.00
B. cylindrica	0.00	0.00	4.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	4.35	0.00	0.00	0.00
C. philippinensis	0.00	4.17	0.00	0.00	0.00	0.00	0.00
C. decandra	0.00	2.08	0.00	**52.17**	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	11.27
N. fruticans	**97.38**	29.17	16.00	0.00	**94.12**	7.69	**87.32**
X. granatum	0.00	6.25	0.00	8.70	0.00	15.38	0.00
X. moluccensis	0.00	4.17	24.00	0.00	0.00	15.38	0.00

b) Relative frequency

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L
A. marina	0.00	5.26	0.00	0.00	0.00	0.00	0.00
A. officinalis	40.00	21.05	28.57	0.00	0.00	0.00	36.36
A. rumphiana	0.00	0.00	0.00	**33.34**	33.34	21.43	0.00
B. cylindrica	0.00	0.00	7.15	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	22.22	0.00	0.00	0.00
C. philippinensis	0.00	5.26	0.00	0.00	0.00	0.00	0.00
C. decandra	0.00	10.53	0.00	22.22	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	9.09
N. fruticans	**60.00**	**31.58**	**42.86**	0.00	**66.66**	**42.86**	**54.54**
X. granatum	0.00	10.53	0.00	22.22	0.00	14.29	0.00
X. moluccensis	0.00	15.79	21.43	0.00	0.00	21.43	0.00

c) Relative dominance

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L
A. marina	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	0.00	**96.95**	**100.00**	0.00	0.00	0.00	**70.27**
A. rumphiana	0.00	0.00	0.00	**98.93**	**100.00**	**99.68**	0.00
B. cylindrica	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	0.24	0.00	0.00	0.00
C. philippinensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. decandra	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	29.73
N. fruticans	0.00	0.00	0.00	0.00	0.00	0.00	0.00
X. granatum	0.00	0.00	0.00	0.83	0.00	0.00	0.00
X. moluccensis	0.00	3.05	0.00	0.00	0.00	0.32	0.00

d) Importance value

Species/Quadrat	1-L	1-R	2-L	2-R	3-L	3-R	4-L
A. marina	0.00	7.35	0.00	0.00	0.00	0.00	0.00
A. officinalis	42.62	**170.08**	**184.57**	0.00	0.00	0.00	108.04
A. rumphiana	0.00	0.00	0.00	**167.05**	139.22	**182.64**	0.00
B. cylindrica	0.00	0.00	11.15	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	26.81	0.00	0.00	0.00
C. philippinensis	0.00	9.43	0.00	0.00	0.00	0.00	0.00
C. decandra	0.00	12.61	0.00	74.40	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	50.10
N. fruticans	**157.38**	60.74	58.86	0.00	**160.78**	50.55	**141.87**
X. granatum	0.00	16.78	0.00	31.74	0.00	29.67	0.00
X. moluccensis	0.00	23.01	45.43	0.00	0.00	37.14	0.00
No. of Species	**2**	**7**	**4**	**4**	**2**	**4**	**3**

officinalis was the most frequent. *Avicennia marina* and *Camptostemon philippinensis*, occupying only one of the seven quadrats (quadrat 1) had the lowest relative frequency values (Table 2.3b). All quadrats were dominated by either *Avicennia officinalis* or *Avicennia rumphiana*, comprising more than 90% of the relative dominance values in each quadrat except for quadrat 4 where *Heritiera littoralis* contributed quite significantly (Table 2.3c).

The importance value of each species in their respective quadrats is shown in Table 2.3d. The most important species were *Avicennia officinalis*, *Avicennia rumphiana* and *Nypa fruticans*. The *Avicennia* spp. were most important in the quadrats on the right while *Nypa fruticans* was most important on the left. The two species of *Xylocarpus* and *Heritiera littoralis* had high importance values in some quadrats.

Species diversity in Creek B (Fig. 2.4b) was higher in the quadrats on the right (mean ± S.E.=0.51 ± 0.03) than on the left (mean ± S.E.=0.36 ± 0.05) with the highest value on the right of quadrat 1 (0.58) and lowest on its left (0.27). There was no correlation between distance and height of river bed with species diversity on both the left and right sides of the creek.

Creek C – Eighteen species of mangroves were sampled in the quadrats in Creek C namely *Aegiceras corniculatum*, *Avicennia alba*, *Avicennia marina*, *Avicennia officinalis*, *Avicennia rumphiana*, *Bruguiera cylindrica*, *Bruguiera gymnorrhiza*, *Bruguiera parviflora*, *Camptostemon philippinensis*, *Ceriops decandra*, *Excoecaria agallocha*, *Heritiera littoralis*, *Nypa fruticans*, *Rhizophora apiculata*, *Rhizophora mucronata*, *Sonneratia alba*, *Xylocarpus granatum* and *Xylocarpus moluccensis* (Table 2.1). Of these 18 species, *Bruguiera parviflora*, *Excoecaria agallocha* and *Rhizophora apiculata* were observed only on the left side of the creek while *Avicennia alba*, *Heritiera littoralis* and *Rhizophora mucronata* were only found on the right.

On the left side of the creek, all 14 quadrats contained *Nypa fruticans*, which had the highest relative density in quadrats 3, 5, J1, 7, J2, 10, 12 and 14. The remaining 6 quadrats were dominated by *Avicennia officinalis* (quadrat 11), *Avicennia rumphiana* (quadrat 13), *Ceriops decandra* (quadrats 4, 8 and 9) and *Sonneratia alba* (quadrat 6).

On the right side of the creek, the first quadrat shows a typical species composition of the seaward zone of the mangroves with *Avicennia marina* and *Sonneratia alba* having the highest relative densities. Up to quadrat 9, relative density is almost evenly distributed among the species with most of the quadrats dominated by the *Avicennia* spp. and *Ceriops decandra.* However, in quadrats 12-15 *Nypa fruticans* had the highest relative density (Table 2.4a). Relative frequency was evenly distributed among the species occupying each quadrat, with *Nypa fruticans* and *Ceriops decandra* being equally frequent in most of the quadrats and *Avicennia* spp. being slightly less frequent. *Xylocarpus granatum* and *Xylocarpus moluccensis* were more frequent in this creek than in Creeks A and B (Table 2.4b). The *Avicennia* spp. were the most dominant with *Avicennia marina* dominating the first five quadrats. Some quadrats with trees belonging to a single species showed 100% relative dominance, like *Avicennia officinalis* (quadrat 12 on the left), *Xylocarpus granatum* (quadrat J1, quadrat 12 on the right) and *Xylocarpus moluccensis* (quadrat 11 on the left) (Table 2.4c).

The importance value of each species in their respective quadrats is shown in Table 2.4d. Highest importance values were shared by the *Avicennia* spp. and *Nypa fruticans* in most of the quadrats, the later being dominant towards the end of Creek C. *Avicennia marina* showed to be dominant in the first five quadrats. Other species which dominated some quadrats were *Xylocarpus* spp. and *Sonneratia alba.*

Highest species diversity on the left side of the creek was 0.88 (quadrat 9) and on the right was 0.90 (quadrat 3) (Fig. 2.4c). On the right side of the creek, a decreasing trend was observed from quadrat 3 to quadrat 14 which had its lowest diversity of 0.46. The lowest diversity recorded on the left side of the creek was on quadrats J1 and 12 (0.30). There was no correlation between distance and height of river bed with species diversity on either side of the creek.

Since there was no significant difference in the diversity between the left and right side of the creek, they were combined for the analysis of diversity between the three creeks. Creek C (0.67 ± 0.04) had significantly higher diversity (ANOVA, $F=5.22$, $p<0.01$) than Creek B (0.43 ± 0.04), but both creeks do not significantly differ from Creek A (0.60 ± 0.05).

Reference for quadrats 1 – 15, J1 and J2 in Creek C

Table 2.4. a) Relative density, b) relative frequency, c) relative dominance, and d) importance value of the mangrove species sampled in the quadrats on the left (L) and right (R) side of Creek C in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines. Values in boldface are the highest in their respective quadrats.

a) Relative density (quadrats 1 to J2)

Species/Quadrat	1-R	2-R	3-L	3-R	4-L	5-L	5-R	J1-L	6-L	6-R	7-L	7-R	J2-L
A. corniculatum	0.00	5.97	0.00	3.17	2.95	0.00	0.67	0.00	1.08	1.44	0.00	0.00	0.00
A. alba	1.92	0.00	0.00	1.59	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	**51.92**	**51.26**	4.20	14.29	28.15	0.00	11.33	0.00	2.69	0.72	3.23	2.38	0.00
A. officinalis	1.92	0.00	0.42	6.35	0.00	0.00	5.33	0.00	1.08	**70.86**	0.00	14.29	0.00
A. rumphiana	0.00	0.00	1.26	7.94	0.00	1.63	0.00	0.00	4.30	1.08	1.61	14.29	0.83
B. cylindrica	0.00	0.00	0.42	1.59	19.84	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41
B. gymnorrhiza	0.00	1.26	0.00	0.00	2.95	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. parviflora	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	0.00	1.57	0.00	0.00	0.00	0.00	0.00	0.00	1.08	2.52	20.97	21.43	0.00
C. decandra	0.00	29.56	11.34	**19.05**	**29.76**	3.25	**46.67**	0.00	10.75	22.30	6.45	**33.33**	2.07
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	**79.83**	0.00	12.87	**95.12**	28.67	**99.69**	27.42	0.00	**54.84**	0.00	**95.85**
R. apiculata	0.00	0.00	0.00	0.00	0.27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	8.81	0.00	**19.05**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	44.23	0.31	2.10	14.29	1.88	0.00	6.00	0.00	**51.61**	0.72	1.61	0.00	0.00
X. granatum	0.00	0.31	0.00	0.00	0.27	0.00	0.00	0.31	0.00	0.36	4.84	11.90	0.00
X. moluccensis	0.00	0.94	0.42	12.70	1.07	0.00	1.33	0.00	0.00	0.00	6.45	2.38	0.83

a) Relative density (quadrats 8 to 15)

Species/Quadrat	8-L	8-R	9-L	9-R	10-L	11-L	12-L	12-R	13-L	14-L	14-R	15-R
A. corniculatum	0.00	0.00	7.69	0.00	1.50	0.00	0.00	0.00	1.49	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	1.74	0.53	2.88	0.00	2.26	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	12.17	22.11	1.92	0.00	23.31	**53.93**	1.10	3.43	11.94	4.43	12.99	0.00
A. rumphiana	2.61	1.58	8.65	**30.11**	0.00	0.00	0.00	12.25	**31.34**	0.00	0.00	22.45
B. cylindrica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.96	0.00	0.00
B. parviflora	0.00	0.00	0.96	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	3.48	5.79	1.92	2.15	3.01	0.00	0.00	0.00	0.00	0.00	0.00	2.04
C. decandra	**34.78**	**37.89**	**40.38**	**30.11**	30.08	2.81	0.00	0.00	17.91	15.82	2.60	0.00
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.27	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.12
N. fruticans	30.43	22.63	24.04	24.73	**32.33**	41.57	**98.90**	**83.82**	26.12	**70.89**	**83.12**	**67.35**
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.00	0.00
X. granatum	2.61	1.58	7.69	4.30	2.26	0.00	0.00	0.49	6.72	0.00	0.00	0.00
X. moluccensis	12.17	7.89	3.85	8.60	5.26	1.69	0.00	0.00	3.73	0.63	1.30	2.04

b) Relative frequency (quadrats 1 to J2)

Species/Quadrat	1-R	2-R	3-L	3-R	4-L	5-L	5-R	J1-L	6-L	6-R	7-L	7-R	J2-L
A. corniculatum	0.00	9.18	0.00	8.04	8.04	0.00	8.33	0.00	7.83	8.33	0.00	0.00	0.00
A. alba	4.65	0.00	0.00	1.79	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	32.56	14.29	11.77	12.50	12.50	0.00	12.96	0.00	12.17	12.96	11.67	13.08	0.00
A. officinalis	**39.53**	0.00	14.29	15.18	0.00	0.00	15.74	0.00	14.78	15.74	0.00	15.89	0.00
A. rumphiana	0.00	0.00	12.61	13.39	0.00	27.28	0.00	0.00	13.04	13.89	12.50	14.02	19.23
B. cylindrica	0.00	0.00	4.20	4.46	4.46	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.41
B. gymnorrhiza	0.00	2.04	0.00	0.00	1.79	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. parviflora	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	0.00	10.20	0.00	0.00	0.00	0.00	0.00	0.00	8.70	9.26	8.33	9.35	0.00
C. decandra	0.00	**20.41**	**16.81**	**17.86**	**17.86**	**36.36**	**18.52**	0.00	**17.39**	**18.52**	**16.67**	**18.69**	**25.64**
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	**16.81**	0.00	**17.86**	**36.36**	**18.52**	**60.61**	**17.39**	0.00	**16.67**	0.00	**25.64**
R. apiculata	0.00	0.00	0.00	0.00	0.89	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	2.04	0.00	1.79	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	23.26	10.20	8.40	8.93	8.93	0.00	9.26	0.00	8.70	9.26	8.33	0.00	0.00
X. granatum	0.00	13.27	0.00	0.00	11.61	0.00	0.00	39.39	0.00	12.04	10.83	12.15	0.00
X. moluccensis	0.00	18.37	15.13	16.07	16.07	0.00	16.67	0.00	0.00	0.00	15.00	16.82	23.08

b) Relative frequency (quadrats 8 to 15)

Species/Quadrat	8-L	8-R	9-L	9-R	10-L	11-L	12-L	12-R	13-L	14-L	14-R	15-R
A. corniculatum	0.00	0.00	6.57	0.00	7.44	0.00	0.00	0.00	7.38	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	11.02	11.02	10.22	0.00	11.57	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	13.39	13.39	12.40	0.00	14.05	22.66	45.95	26.16	13.94	21.79	22.66	0.00
A. rumphiana	11.81	11.81	10.95	15.62	0.00	0.00	0.00	23.07	12.29	0.00	0.00	23.44
B. cylindrica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.56	0.00	0.00
B. parviflora	0.00	0.00	0.73	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	7.87	7.87	7.30	10.43	8.26	0.00	0.00	0.00	0.00	0.00	0.00	15.63
C. decandra	**15.75**	**15.75**	**14.60**	**20.83**	**16.53**	**26.67**	0.00	0.00	**16.39**	**25.64**	**26.67**	0.00
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.28	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.56
N. fruticans	**15.75**	**15.75**	**14.60**	**20.83**	**16.53**	**26.67**	**54.05**	**30.77**	**16.39**	**25.64**	**26.67**	**31.25**
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	8.20	0.00	0.00	0.00
X. granatum	10.24	10.24	9.49	13.54	10.74	0.00	0.00	20.00	10.66	0.00	0.00	0.00
X. moluccensis	14.17	14.17	13.14	18.75	14.88	24.00	0.00	0.00	14.75	23.09	24.00	28.12

c) Relative dominance (quadrats 1 to J2)

Species/Quadrat	1-R	2-R	3-L	3-R	4-L	5-L	5-R	J1-L	6-L	6-R	7-L	7-R	J2-L
A. corniculatum	0.00	0.00	0.00	1.27	2.47	0.00	1.52	0.00	3.46	0.92	0.00	0.00	0.00
A. alba	3.85	0.00	0.00	0.75	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	**73.09**	**85.82**	**78.01**	**42.32**	**70.13**	0.00	**59.62**	0.00	22.36	12.73	**64.56**	9.23	0.00
A. officinalis	0.00	0.00	0.00	0.59	0.00	0.00	20.06	0.00	0.00	9.02	0.00	0.00	0.00
A. rumphiana	0.00	0.00	7.58	3.66	0.00	0.00	0.00	0.00	23.40	33.66	0.00	**47.22**	**78.78**
B. cylindrica	0.00	0.00	3.81	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	0.00	18.36	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. parviflora	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.64	1.67	18.61	29.97	0.00
C. decandra	0.00	0.00	0.00	0.00	1.15	0.00	0.00	0.00	9.15	0.47	1.79	0.21	0.00
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. apiculata	0.00	0.00	0.00	0.00	1.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	10.57	0.00	26.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	23.07	3.61	10.60	25.30	6.74	0.00	18.81	0.00	**36.99**	**41.53**	9.82	0.00	0.00
X. granatum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	**100.00**	0.00	0.00	3.62	11.75	0.00
X. moluccensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.60	1.62	21.22

c) Relative dominance (quadrats 8 to 15)

Species/Quadrat	8-L	8-R	9-L	9-R	10-L	11-L	12-L	12-R	13-L	14-L	14-R	15-R
A. corniculatum	0.00	0.00	3.27	0.00	1.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	29.58	19.51	41.16	0.00	**49.23**	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	**48.96**	0.00	1.27	0.00	14.69	0.00	**100.00**	0.00	3.20	**51.44**	0.00	0.00
A. rumphiana	0.00	**74.36**	**50.10**	**87.95**	0.00	0.00	0.00	0.00	**84.74**	0.00	0.00	**88.96**
B. cylindrica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. parviflora	0.00	0.00	0.47	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	16.01	3.15	0.00	5.38	1.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. decandra	1.17	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00	4.78	0.00	0.00
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	26.15	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.85	0.00	0.00	0.00
X. granatum	4.29	1.90	3.48	3.07	2.08	0.00	0.00	**100.00**	4.05	0.00	0.00	0.00
X. moluccensis	0.00	1.09	0.00	3.60	31.72	**100.00**	0.00	0.00	7.17	17.63	0.00	11.04

d) Importance value (quadrats 1 to J2)

Species/Quadrat	1-R	2-R	3-L	3-R	4-L	5-L	5-R	J1-L	6-L	6-R	7-L	7-R	J2-L
A. corniculatum	0.00	15.16	0.00	12.48	13.45	0.00	10.52	0.00	12.36	10.68	0.00	0.00	0.00
A. alba	10.42	0.00	0.00	4.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	**157.57**	**151.36**	93.97	**69.10**	**110.78**	0.00	**83.91**	0.00	37.22	26.41	**79.45**	24.69	0.00
A. officinalis	41.46	0.00	14.71	22.12	0.00	0.00	41.13	0.00	15.86	**95.62**	0.00	30.18	0.00
A. rumphiana	0.00	0.00	21.45	24.99	0.00	28.91	0.00	0.00	40.74	48.63	14.11	**75.53**	98.84
B. cylindrica	0.00	0.00	8.44	6.05	24.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.82
B. gymnorrhiza	0.00	3.30	0.00	0.00	23.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. parviflora	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	0.00	11.78	0.00	0.00	0.00	0.00	0.00	0.00	14.42	13.45	47.91	60.75	0.00
C. decandra	0.00	49.97	28.15	36.90	48.77	39.61	65.19	0.00	37.29	41.29	24.91	52.23	27.71
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. fruticans	0.00	0.00	**96.64**	0.00	30.72	**131.48**	47.19	**160.30**	44.81	0.00	71.51	0.00	**121.49**
R. apiculata	0.00	0.00	0.00	0.00	2.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	21.42	0.00	46.94	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	90.55	14.13	21.10	48.52	17.55	0.00	34.07	0.00	**97.30**	51.51	19.76	0.00	0.00
X. granatum	0.00	13.58	0.00	0.00	11.87	0.00	0.00	139.70	0.00	12.40	19.29	35.80	0.00
X. moluccensis	0.00	19.31	15.55	28.77	17.14	0.00	18.00	0.00	0.00	0.00	23.05	20.82	45.13
No. of Species	**4**	**9**	**8**	**10**	**10**	**3**	**7**	**2**	**8**	**8**	**8**	**7**	**5**

d) Importance value (quadrats 8 to 15)

Species/Quadrat	8-L	8-R	9-L	9-R	10-L	11-L	12-L	12-R	13-L	14-L	14-R	15-R
A. corniculatum	0.00	0.00	17.53	0.00	10.19	0.00	0.00	0.00	8.87	0.00	0.00	0.00
A. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. marina	42.34	31.06	54.26	0.00	**63.05**	0.00	0.00	0.00	0.00	0.00	0.00	0.00
A. officinalis	**74.52**	35.50	15.59	0.00	52.04	76.59	147.05	29.59	29.08	77.66	35.65	0.00
A. rumphiana	14.42	**87.75**	**69.70**	**133.68**	0.00	0.00	0.00	35.32	**128.37**	0.00	0.00	**134.85**
B. cylindrica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
B. gymnorrhiza	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	9.52	0.00	0.00
B. parviflora	0.00	0.00	2.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. philippinensis	27.36	16.81	9.22	17.96	12.31	0.00	0.00	0.00	0.00	0.00	0.00	17.67
C. decandra	51.70	53.64	55.24	50.94	46.61	29.48	0.00	0.00	34.30	46.24	29.27	0.00
E. agallocha	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	28.69	0.00	0.00
H. littoralis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	7.68
N. fruticans	46.18	38.38	38.64	45.56	48.86	68.24	**152.95**	114.59	42.51	**96.53**	**109.79**	98.60
R. apiculata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R. mucronata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S. alba	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	9.80	0.00	0.00	0.00
X. granatum	17.14	13.72	20.67	20.91	15.08	0.00	0.00	**120.49**	21.42	0.00	0.00	0.00
X. moluccensis	26.34	23.15	16.99	30.95	51.86	**125.69**	0.00	0.00	25.65	41.35	25.30	41.20
No. of Species	**8**	**8**	**10**	**6**	**8**	**4**	**2**	**4**	**8**	**6**	**4**	**5**

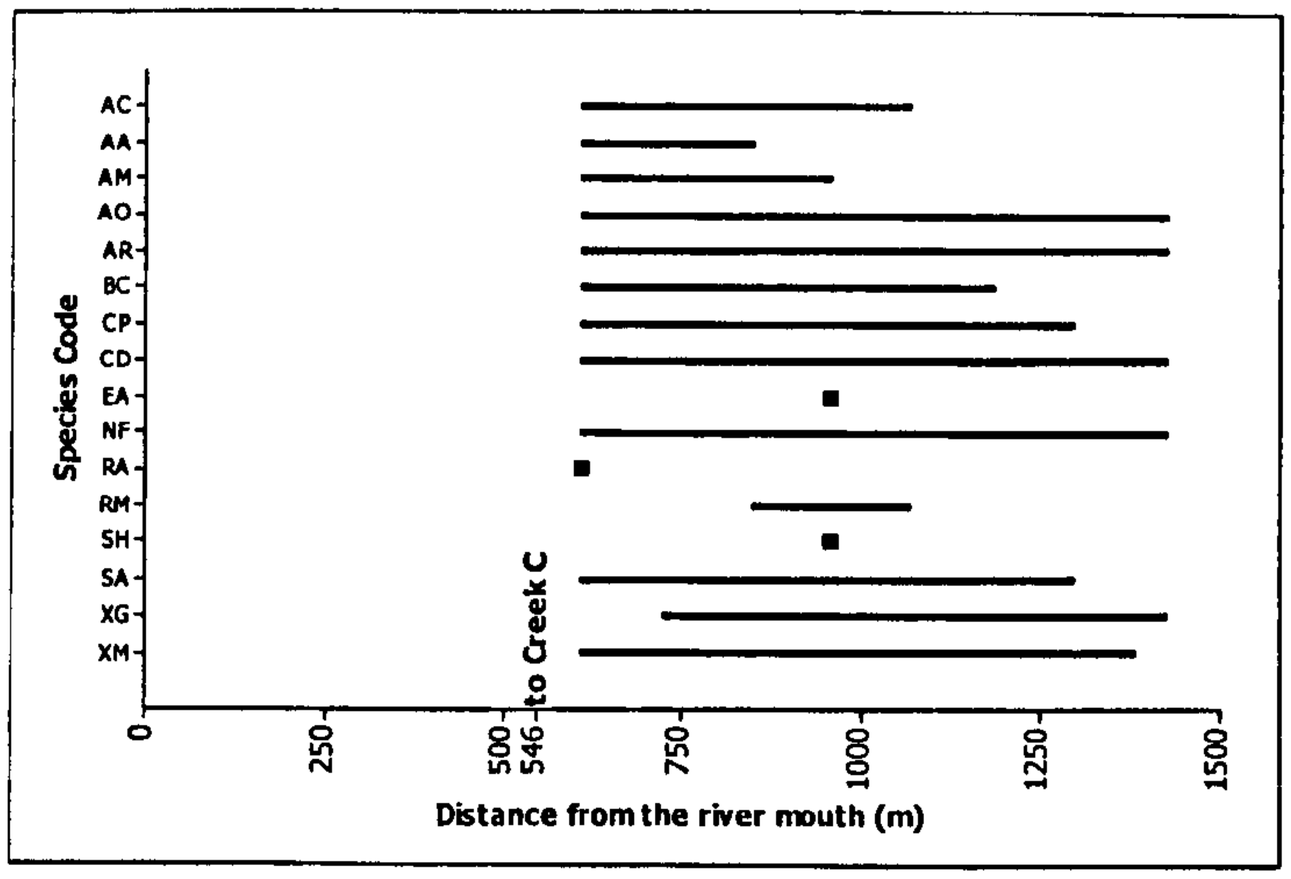
Although no clear zonation pattern was observed along the creeks, some patterns of species distribution were noted (Fig. 2.5). The longest creek is Creek C (2,018 m). The first quadrat was laid 353 m away from the river mouth on Creek C. Only four species were observed in this quadrat, *Avicennia alba*, *Avicennia marina*, *Avicennia officinalis* and *Sonneratia alba*. Of these four species, *Avicennia alba* had the most limited distribution extending only up to 850 m from the mouth of the river while *Avicennia officinalis* had the widest range extending up to 1,918 m. The farthest limit for *Avicennia marina* and *Sonneratia alba* were 1,515 m and 1,800 m, respectively. Among the species with wide distribution upstream were *Aegiceras corniculatum* (477-1,800 m), *Avicennia rumphiana* (604-2,018 m), *Bruguiera cylindrica* (604-2,018 m), *Bruguiera gymnorrhiza* (477-1,918 m), *Camptostemon philippinensis* (477-2,018 m), *Ceriops decandra* (477-1,918 m), *Nypa fruticans* (604-2,018 m), *Xylocarpus granatum* (477-1,800 m) and *Xylocarpus moluccensis* (477-2,018 m). *Bruguiera parviflora*, *Excoecaria agallocha*, *Heritiera littoralis* and *Scyphiphora hydrophyllacea* were found some distance away from the sea and were observed at least 1 km upstream. *Rhizophora apiculata* was observed only between 608-722 m while *Rhizophora mucronata* had a wider distribution (477-1,070 m).

Topographical profile

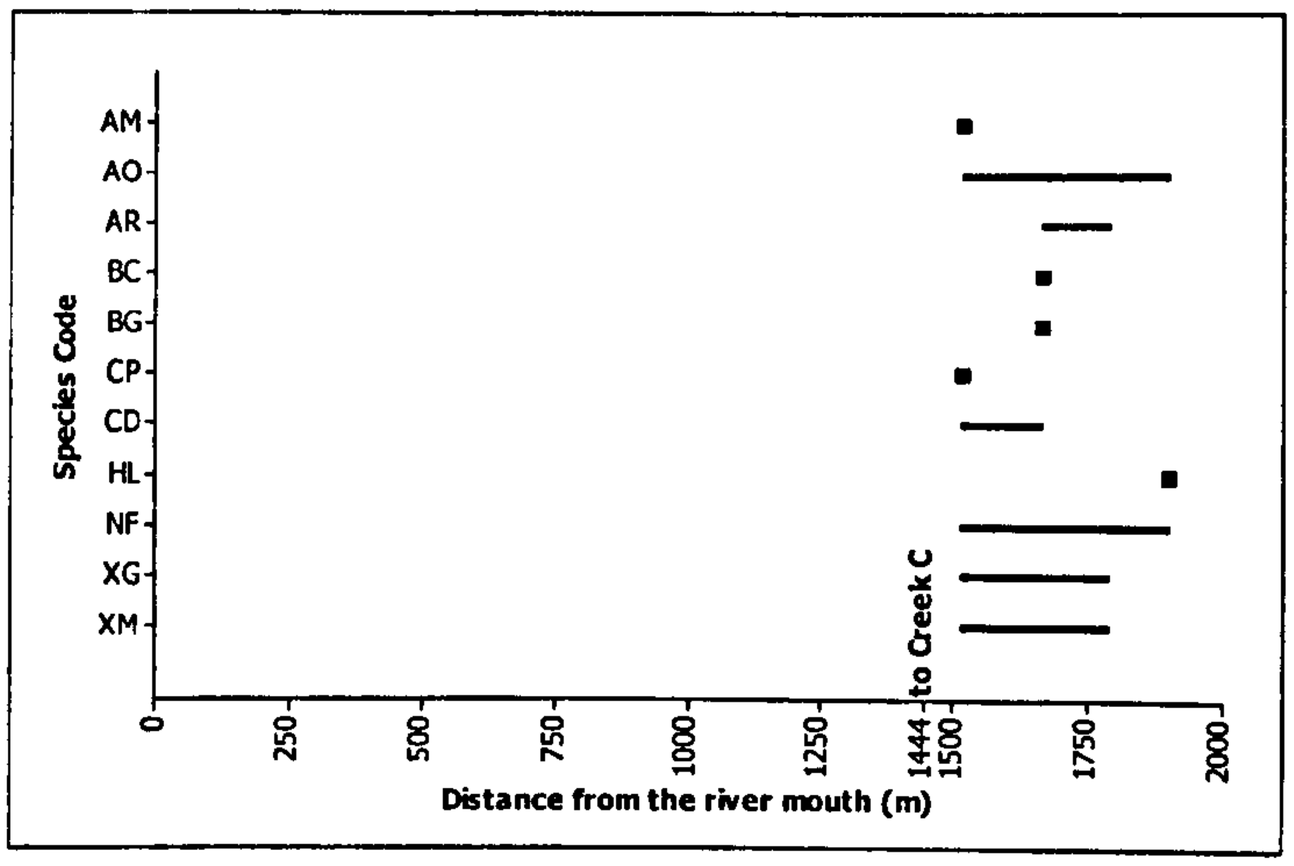
The Naisud River is the only access of the mangroves in Naisud and Bugtong Bato to the Sibuyan Sea. It has two branches, which are referred to in this present work as Creek A and Creek C. Creek C is considered as the main creek because it is the larger and longer of the two creeks. Creek A joins Naisud River at 546 m from the river mouth, while Creek C joins at 723 m. Creek B, on the other hand, joins Creek C at 1,444 m from the river mouth. Creek A supplies water to the mangroves in the Naisud area while Creek C supplies water to the mangroves in Bugtong Bato (Fig. 2.2).

Creek A – From the junction where Creek A joins the Naisud River, it stretches 879 m upstream near the large ponds and the Aklan Highway. From the river mouth, the creek measures 1,425 m in length. The creek bed slopes upward from the junction of Naisud River towards its tip. The heights of the river bank above Chart Datum did not vary much, except where quadrat J3 was located, making the bank lower towards the tip of the creek (Fig. 2.6).

a) Creek A



b) Creek B



c) Creek C

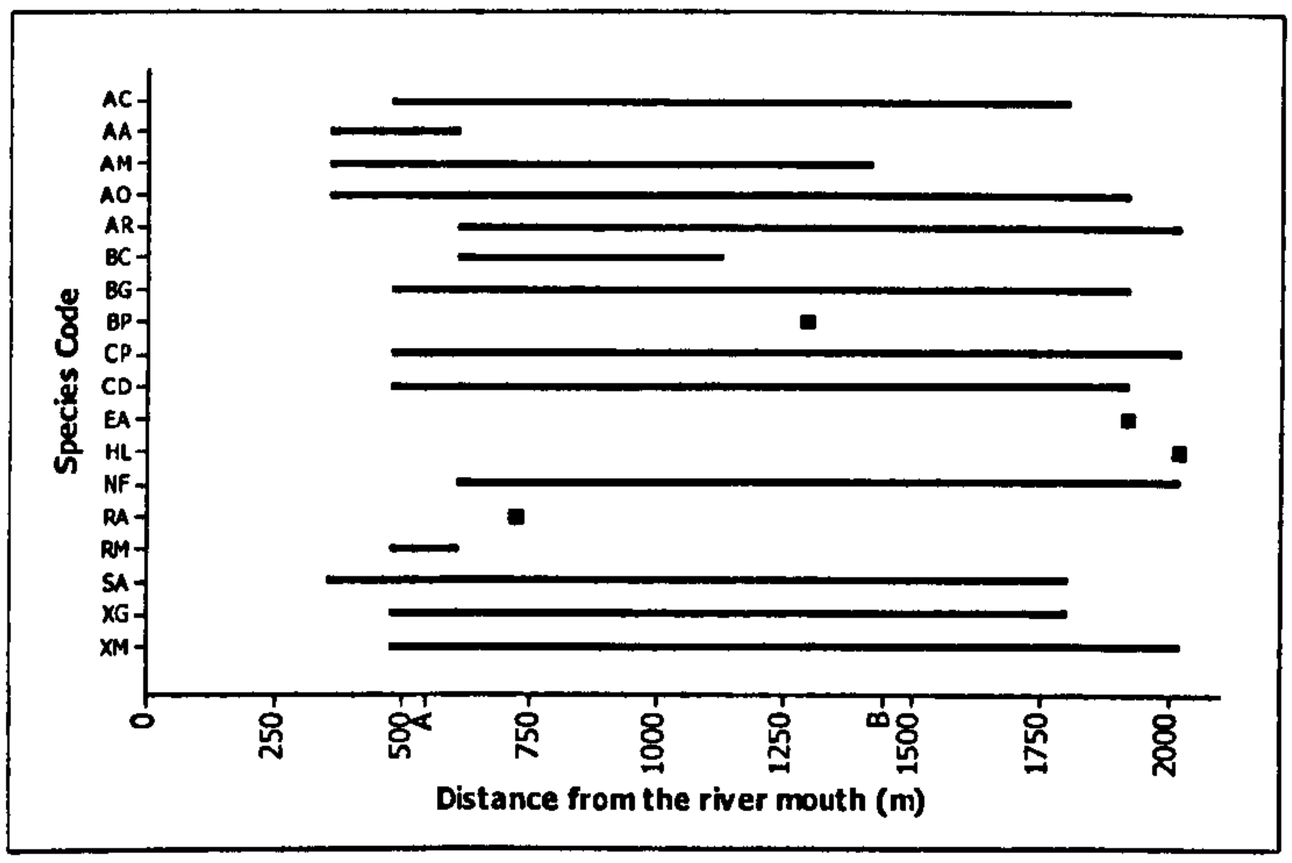
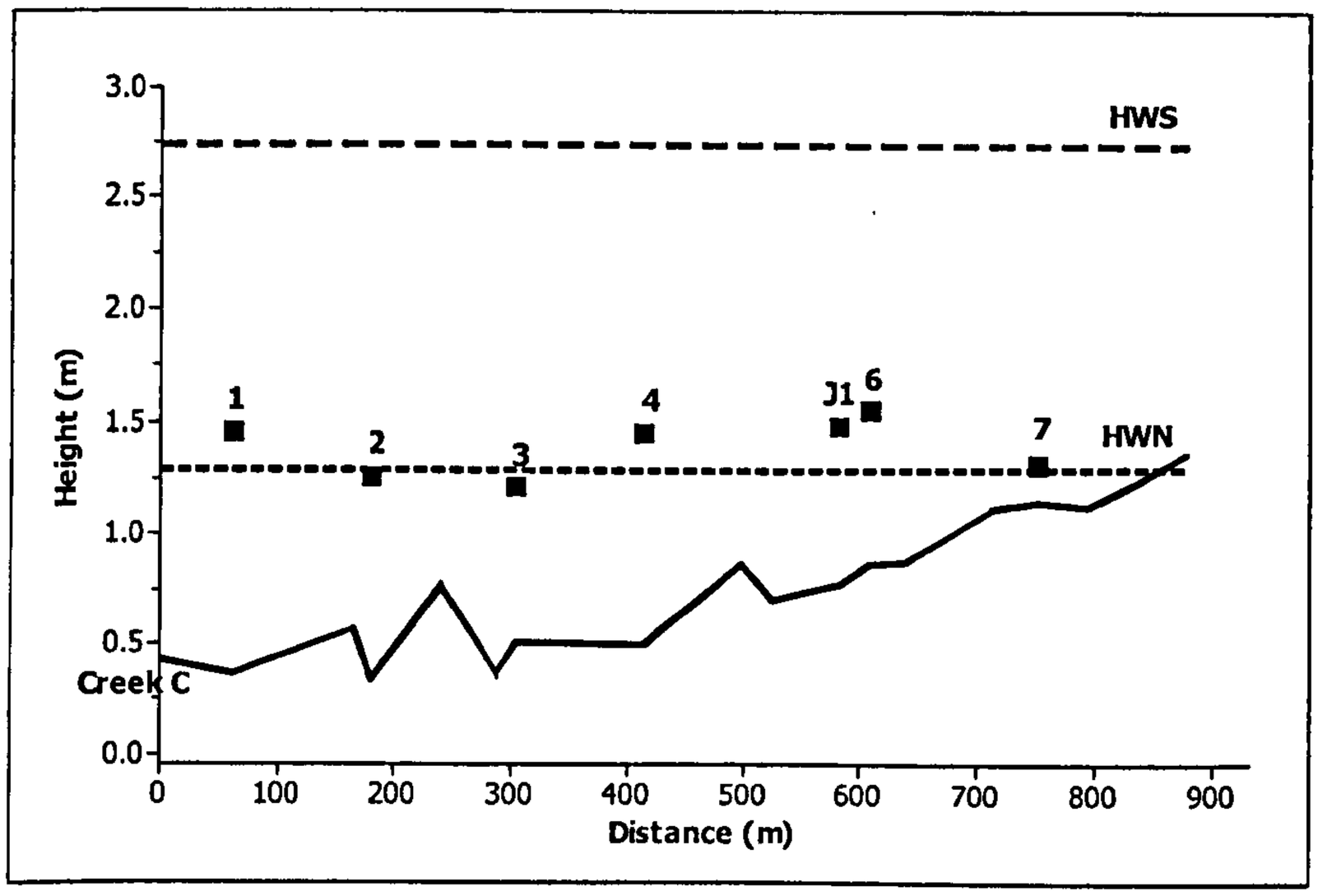


Figure 2.5. Distribution of mangrove species along three creeks of Naisud River, Ibajay, Aklan, Philippines. Distances from the mouth of the river (0 m) where Creeks A and B join Creek C are marked; species codes are defined on the facing page.

a) Left side of the creek



b) Right side of the creek

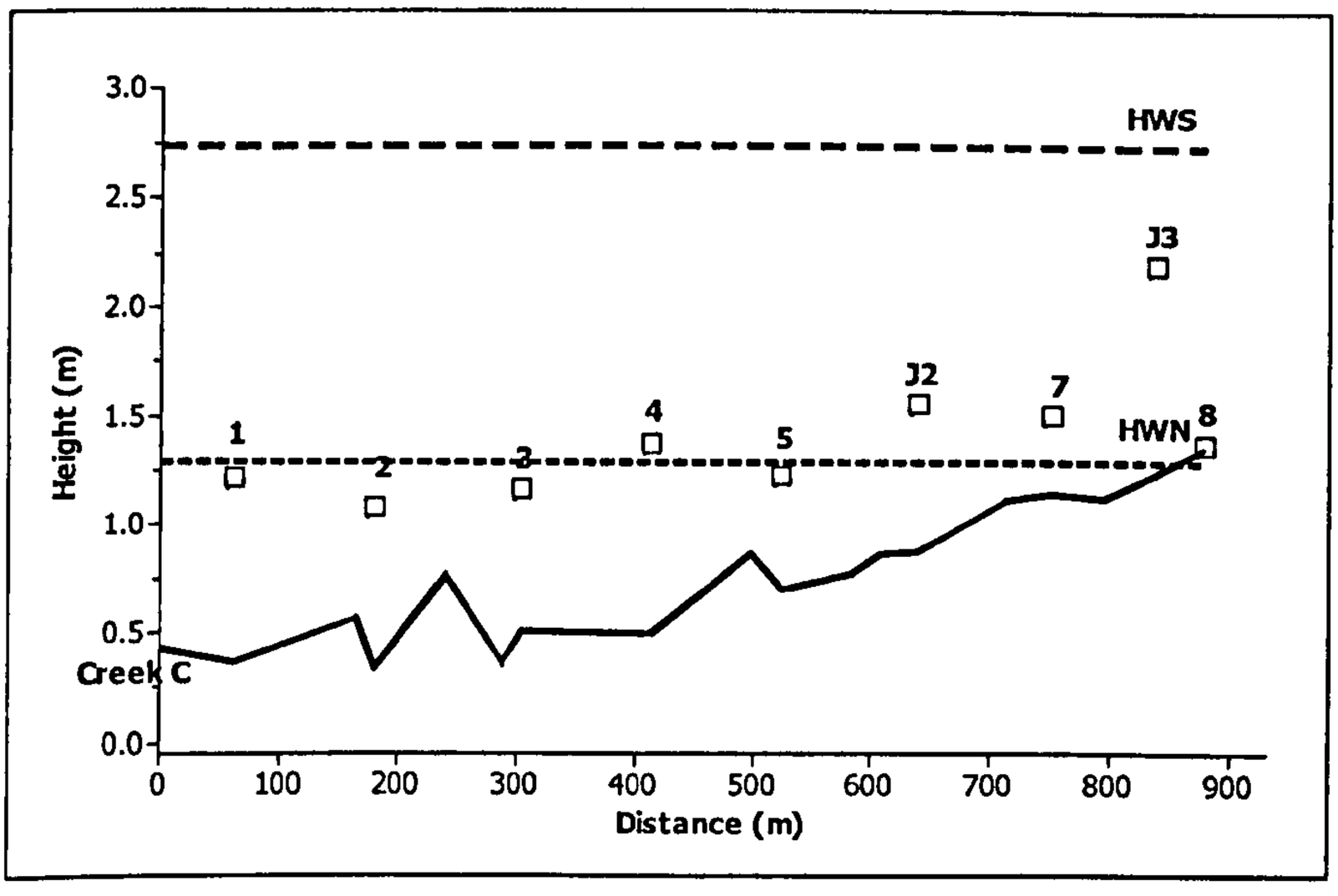


Figure 2.6. Topographical profile of Creek A showing the height of the creek with reference to the Chart Datum (solid line), the height of high water spring (HWS) and high water neap (HWN) and the heights of the banks where the quadrats (numbered symbols) for the mangrove community analysis were laid out both on the a) left (filled squares) and b) right (empty squares) sides of the creek. The part of the creek that joins to Creek C is marked.

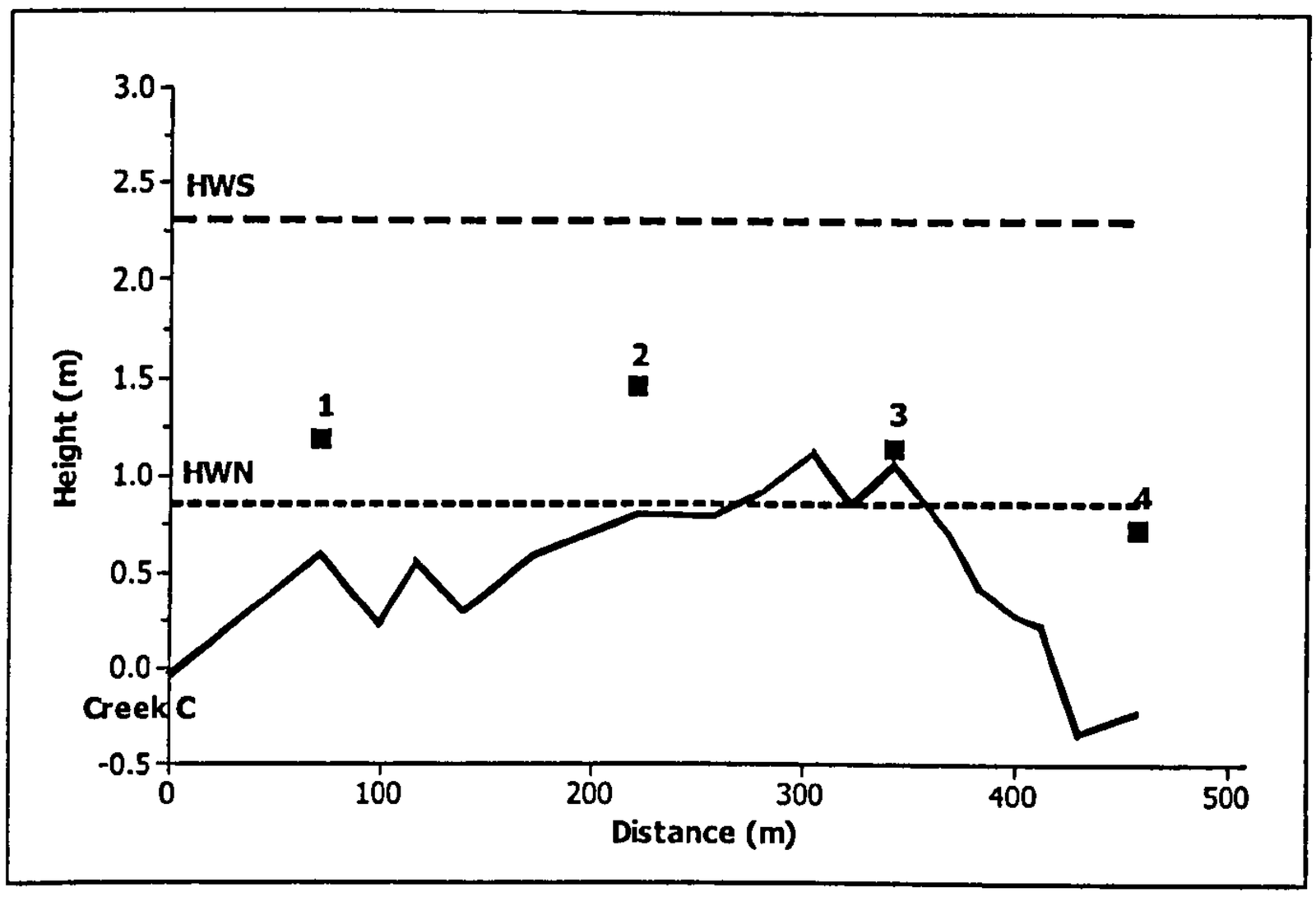
Creek B – From Creek C, Creek B stretches 458 m landward where it ends. From the river mouth to its end, the creek measures 1,902 m. The point where Creek B joins Creek C is the deepest part of the main creek. From this junction the creek slopes upward and peaks 300 m away from the junction then slopes down to 430 m which is -0.33 m below Chart Datum, the deepest portion in the whole mangrove area. The heights of the bank above Chart Datum did not vary much so it became shallower in that portion of the creek where the creek bed sloped upward (Fig. 2.7).

Creek C – Creek C joins the Naisud River at 723 m from the river mouth and stretches 1,296 m upstream. Its total distance from its tip to the river mouth is 2,018 m. It ends beside the dirt road that borders the edge of the mangroves in the east. The creek forms a basin from 605 m to 1,683 m covering a deeper stretch of approximately 1 km. At 605 m the creek bed measuring 0.63 m above Chart Datum started to slope down and sloped upwards again to 0.78 m above Chart Datum at 1,683 m. Between these two points, highs and lows were observed, the lowest being -0.04 m. The heights of the banks above Chart Datum did not vary much except towards the last 4 quadrats where they tended to become lower (Fig. 2.8).

Environmental parameters

Mean water temperature and salinity in the area measured during the duration of the crab sampling was 28.7 ± 0.24°C and 22.6 ± 1.42 p.s.u., respectively. Mean pore water salinity obtained from the quadrats in the present work were 35.6 ± 0.60 in Creek A, 20.4 ± 1.30 in Creek B and 36.9 ± 0.83 in Creek C. Creeks A and C had significantly higher salinity than Creek B (ANOVA, $F=63.77$, $p<0.001$). No correlation was observed between salinity and height of banks above Chart Datum in all three creeks. Mean soil pH obtained from the quadrats in the present work were 6.65 ± 0.08 in Creek A, 6.61 ± 0.08 in Creek B and 6.76 ± 0.05 in Creek C. pH in all three creeks did not significantly differ from each other (ANOVA, $F=1.07$, $p>0.05$). No correlation was observed between pH and height of banks above Chart Datum in all three creeks. Percent soil composition in terms of sand, silt and clay showed a decreasing proportion in all three creeks (Fig. 2.9). There was no significant difference in the silt (ANOVA, $F=2.42$, $p>0.05$), sand ($F=2.46$, $p>0.05$) and clay ($F=1.67$, $p>0.05$) components of the soil between the three creeks.

a) Left side of the creek



b) Right side of the creek

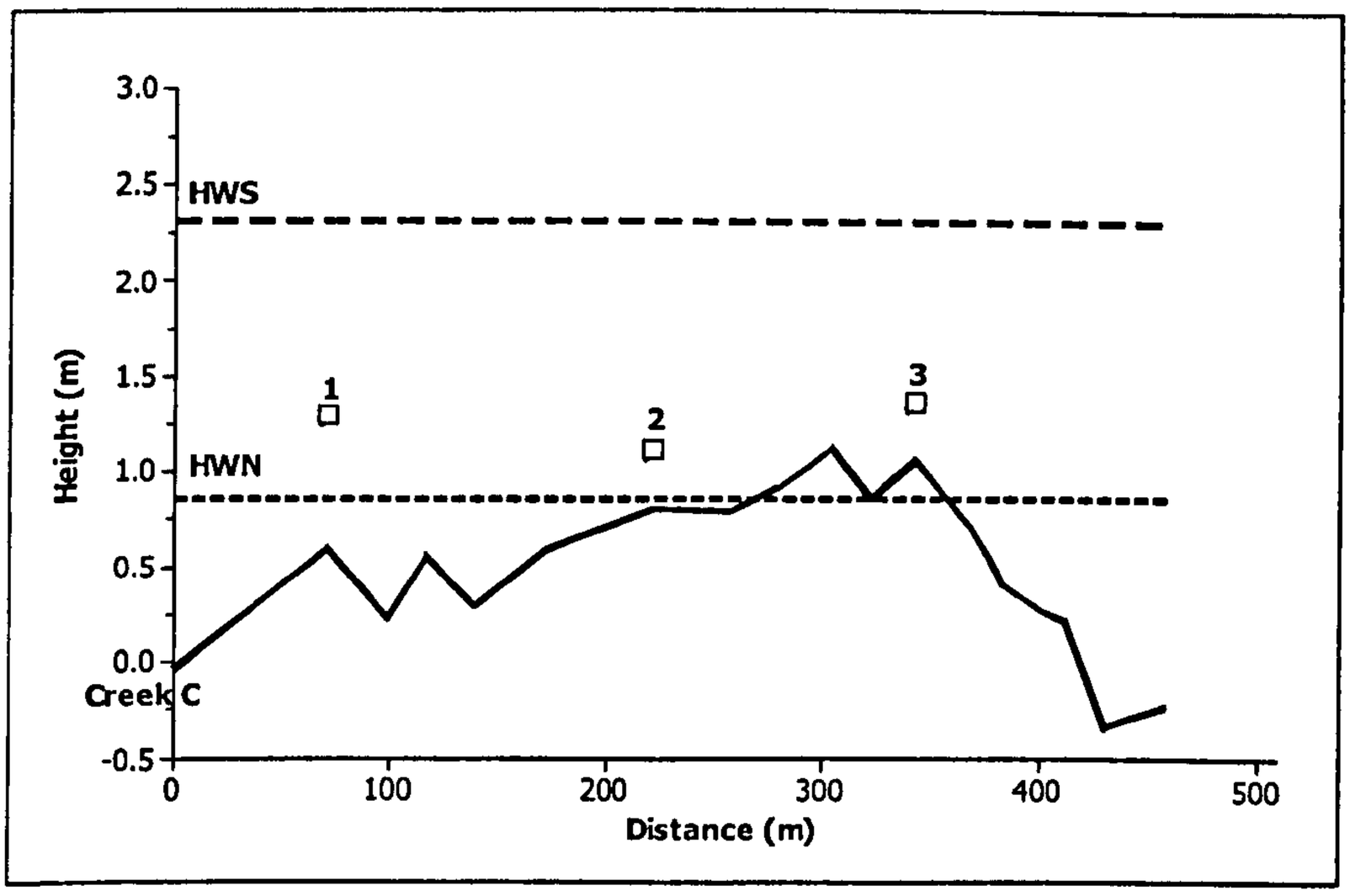
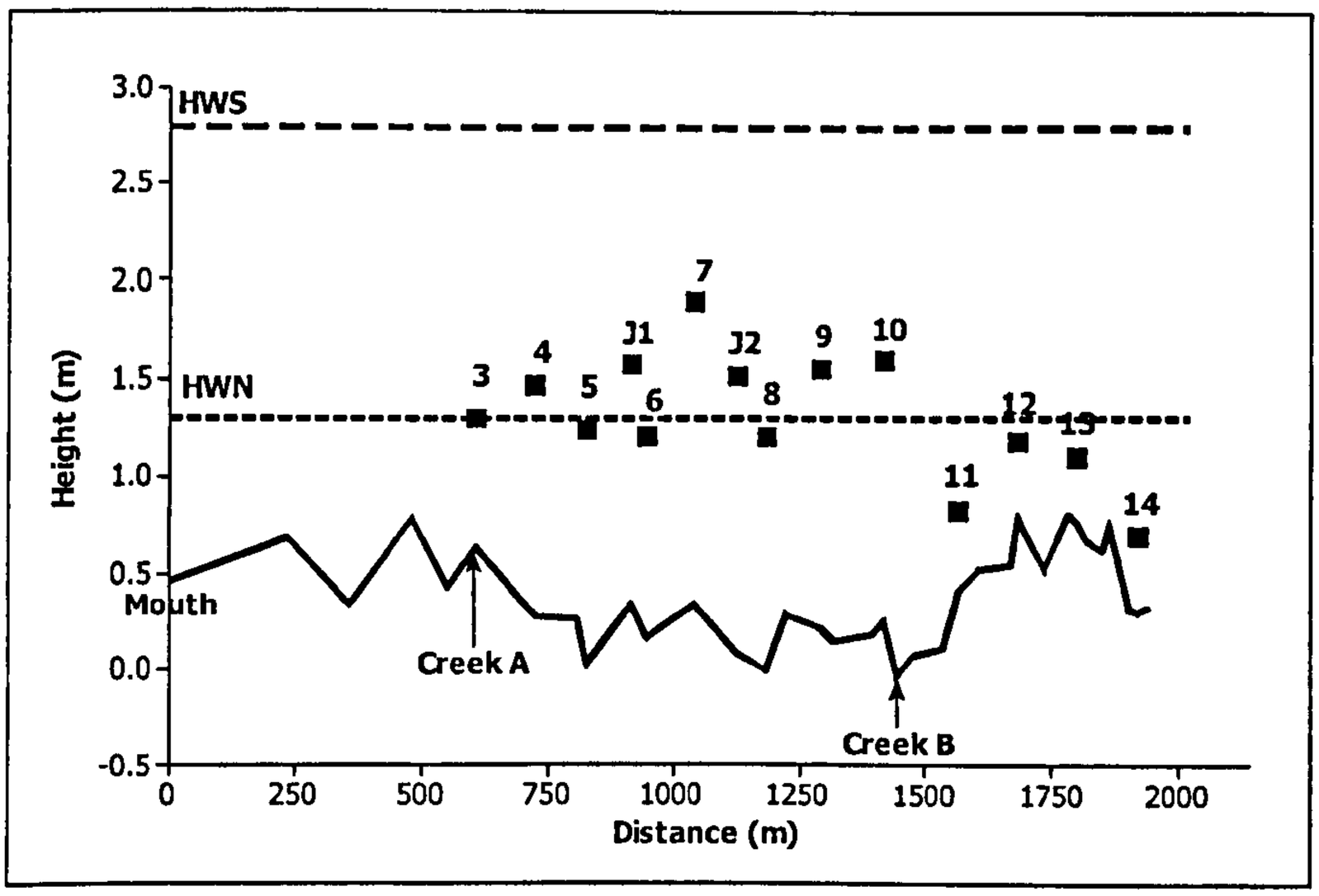


Figure 2.7. Topographical profile of Creek B showing the height of the creek with reference to the Chart Datum (solid line), the height of high water spring (HWS) and high water neap (HWN) and the heights of the banks where the quadrats (numbered symbols) for the mangrove community analysis were laid out both on the a) left (filled squares) and b) right (empty squares) sides of the creek. The part of the creek that joins to Creek C is marked.

a) Left side of the creek



b) Right side of the creek

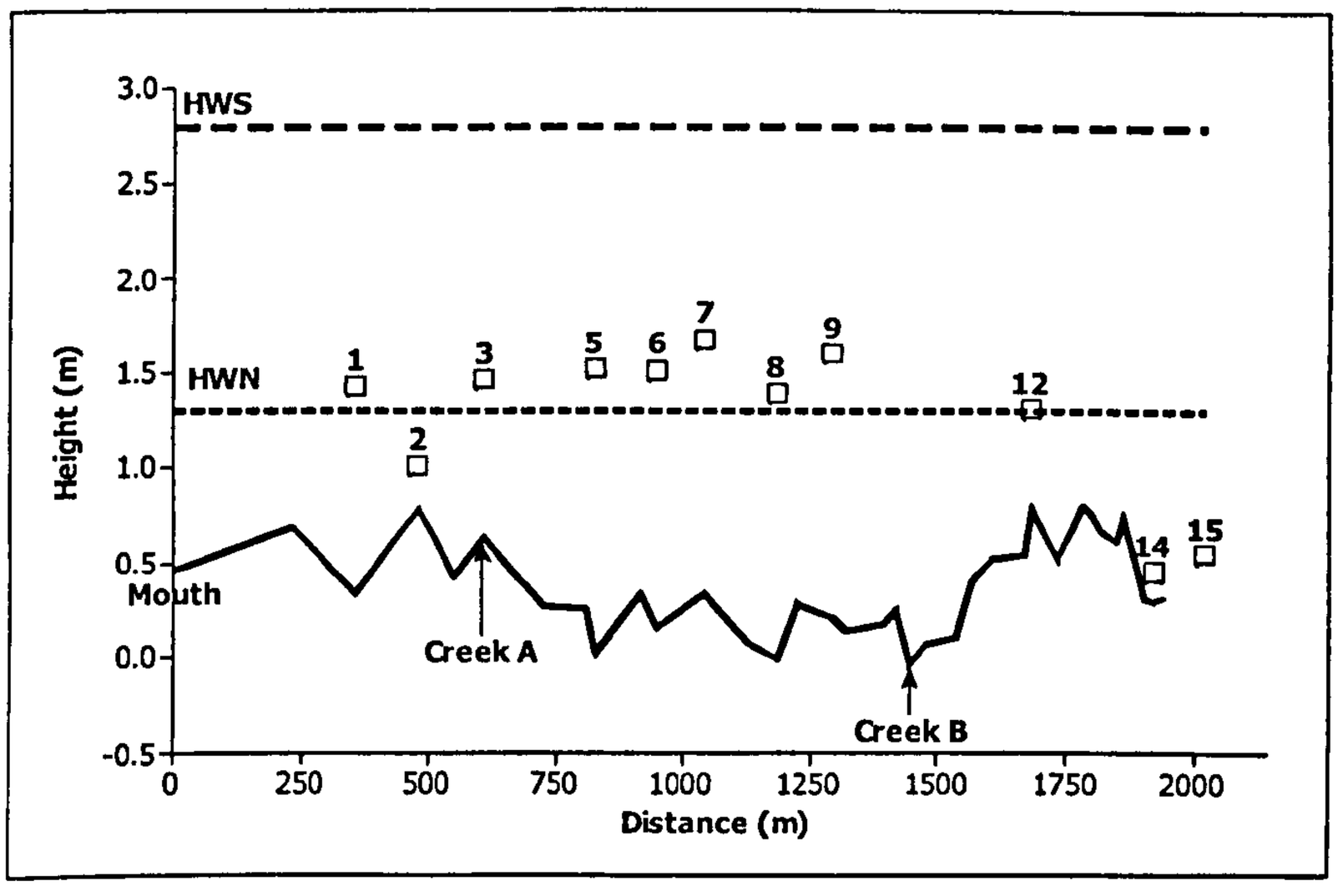


Figure 2.8. Topographical profile of Creek C showing the height of the creek with reference to the Chart Datum (solid line), the height of high water spring (HWS) and high water neap (HWN) and the heights of the banks where the quadrats (numbered symbols) for the mangrove community analysis were laid out both on the a) left (filled squares) and b) right (empty squares) sides of the creek. The parts of the creek where Creeks A and B join are marked.

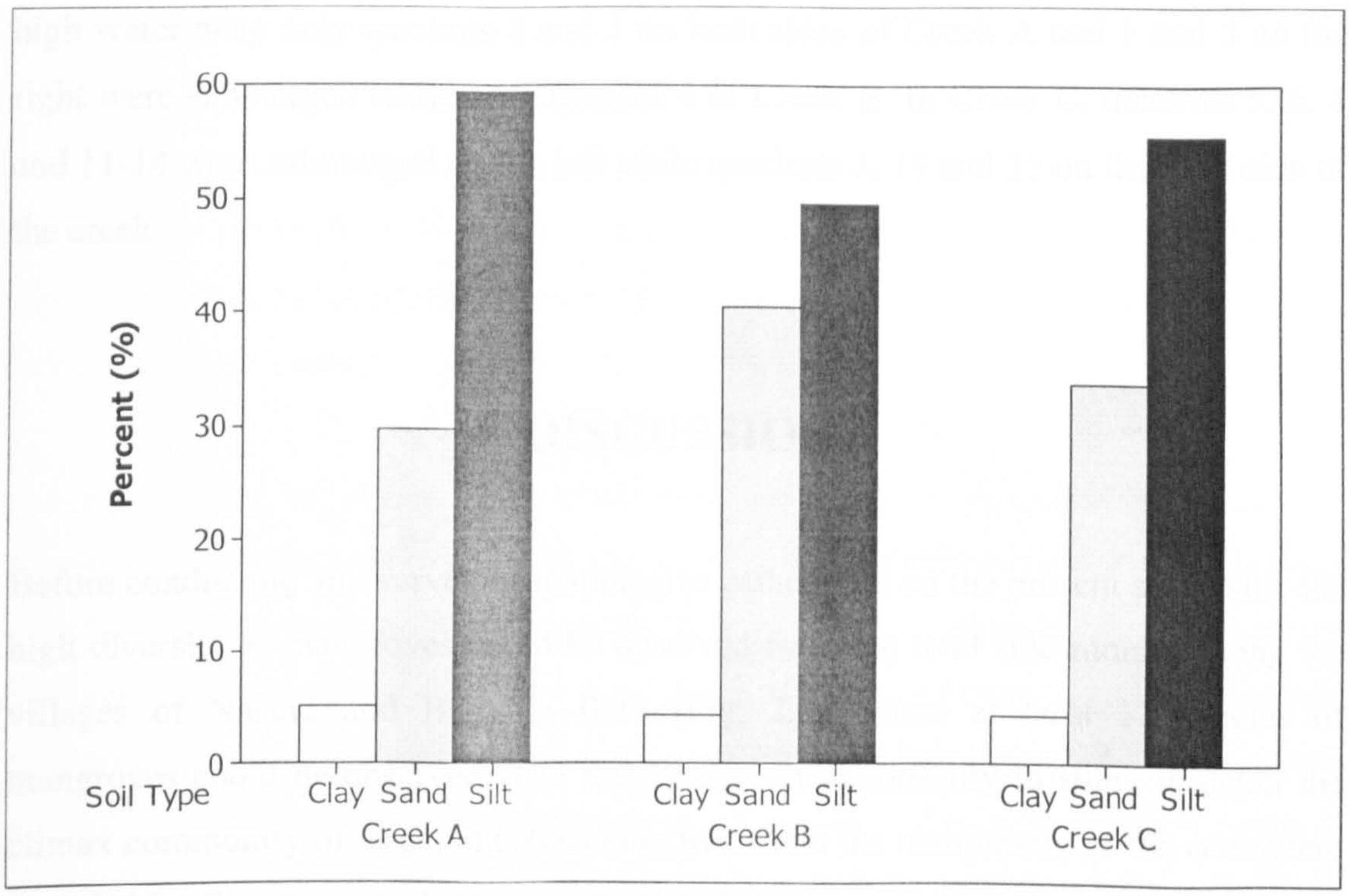


Figure 2.9. Clay, sand and silt composition (%) of the soil in the three creeks sampled in Naisud and Bugtong Bato, Ibajay, Aklan, Philippines.

Tidal inundation

High water spring levels obtained during the highest spring tide of the year were 2.74 m in Creek A (Fig. 2.6), 2.31 m in Creek B (Fig. 2.7) and 2.8 m in Creek C (Fig. 2.8) while the predicted tide height during that time was 2.3 m (2005 Tide Table). High water neap estimates were obtained from the predicted height in the tide table adjusted with the error obtained when taking measurements for high water spring. High water neap levels were 1.29 m in Creek A, 0.86 m in Creek B and 1.30 m in Creek C. All quadrats in all creeks were submerged during high water spring. During high water neap only quadrats 2 and 3 on both sides of Creek A and 1 and 5 on the right were submerged while only quadrat 4 in Creek B. In Creek C, quadrats 5, 6, 8 and 11-14 were submerged on the left while quadrats 2, 14 and 15 on the right side of the creek.

DISCUSSION

Before conducting any survey or mapping to gather data on the present study site, the high diversity of mangroves could be observed from the road side running along the villages of Naisud and Bugtong Bato (Fig. 2.2) where at least 12 species of mangroves could be observed. The survey was an opportunity to study in detail the climax community of large, old *Avicennia* trees, and the complexity of the ecosystem revealed by the various adaptations to a harsh environment these plants possess: the complicated root system, salt crystals extruded by the salt glands on the leaves and the unique features of the mangrove propagules. Lobster mounds were widespread in the climax community where a lush growth of *Avicennia* spp. was found. These lobsters may be responsible for the healthy growth of these trees. According to Smith et al. (1991), burrowing crustaceans promote growth of established trees by improving soil aeration and reducing levels of harmful sulphides.

Mapping the area revealed a single narrow entrance from the sea to Naisud River. Along the shoreline where the Naisud River drains, no mangroves can be observed. The isolation of mangroves in the upper reaches of the river may be due to strong

wave action along the coast which is more prevalent during the northeast monsoon (November-April). This may have prevented settlement of mangroves along this area. According to Macnae (1968), mangroves grow only on sheltered shores protected by coral reefs that break the force of waves. Moreover, the sandy substrate along the coast of Naisud and Bugtong Bato may not be suitable for mangroves to settle and grow. Most mangrove species grow best in loose, fine-textured mud or silt, rich in humus (Kathiresan and Bingham 2001); a typical mangrove habitat is a muddy river estuary (Hogarth 1999). Tomlinson (1994) described the mangrove substrate as usually firm to soft mud, and soil type has a major impact on mangrove growth. Hatton and Couto (1992) reported that the movement of sand into the mangroves on Portuguese Island, Mozambique has caused high mortality of *Ceriops tagal*. According to Thom (1984), mangroves commonly occur behind a protective sand barrier. The sandy shore of Naisud has isolated the mangroves from direct contact to the sea and acts as barrier that protects them from direct exposure to strong waves on this high energy coast.

During mapping and survey of the study area, aside from the general observation as to the location of the mangroves and the species that inhabit the community, among other things observed were the aquaculture ponds owned by politicians both of the past and the present and the *Nypa* plantations in the mangroves claimed as private properties by some so-called "owners." Considering the potential of the mangrove habitat as site for aquaculture ponds, some government officials have built ponds in this public land knowing that local fishers or villagers have no power to prevent them. According to Hutchinson (2005), it becomes practically difficult to manage the mangroves when government officials are those owning majority of the ponds within the mangroves. Another widespread mechanism by which mangroves have been lost from the public domain is through payment of real estate tax for these areas claimed as private properties by the "owners" who allegedly inherited portions of these mangroves from their ancestors. The local government, hard-pressed for cash, accept the taxes without checking the real status of the land (Primavera 2000). This system caused the conversion of some portions of the mangroves into *Nypa* plantations and culture ponds and these practises are still going on.

Mangrove community structure

In spite of numerous papers on mangrove floristics, systematics, phytogeography and related topics, there is little published information on the mangrove community structure (Cintron and Novelli 1984). The complex architecture of the forest may be one reason why this aspect of mangrove ecology is less studied. Doing community structure analysis of the mangroves means walking in varying depths of mud, climbing up or crawling under their complicated root systems and working for long hours amidst mosquitoes and other stinging insects. Mud, methane and mosquitoes are features of mangroves, as are snakes (Hogarth 1999). These difficulties did not discourage the conduct of community structure analysis at the present study site where the need was urgent, as there is no baseline evidence to support the observation that during the past decade the area has degenerated remarkably due to anthropogenic disturbances. According to Ashton et al. (2003), such disturbances often result in simplification of the natural ecosystems and diversity loss, but the consequences on ecosystem functioning are often unrecorded due to a lack of baseline studies and monitoring of natural ecosystems. This has been observed in the mangroves of Naisud and Bugtong Bato where some stands of *Scyphiphora hydrophyllacea* had been cut to give way to the expanding fish pond and lately as *Nypa fruticans* plantation continues to expand, it is expected that more and more trees will be felled. The very few remaining stands of *Bruguiera gymnorrhiza*, *Bruguiera parviflora* and *Scyphiphora hydrophyllacea*, some of the rare mangrove species in the present study site, may disappear in a few years if these human activities continue. Hence, the need for immediate documentation for future reference.

The high relative density values of *Nypa fruticans* and *Ceriops decandra* is notable in all three creeks sampled. High density values of *Nypa fruticans* were due to planted *Nypa* that ranged from small patches sprouting all over the area to a few hectares of plantation. *Ceriops decandra* was also dense because of the colonizing nature of the species. This species was observed to be successful colonizers being commonly found growing in abandoned ponds and other cleared areas. According to Tomlinson (1994), *Ceriops decandra* is a typical constituent of the inner mangroves, often forming pure stands on better drained areas. The mangrove study site is drained most of the time, usually twice daily as the tide in the area is diurnal and some parts are not even

flooded during neap tides (Figs. 2.6, 2.7, 2.8). The *Avicennia* trees have relatively high dominance values, while *Nypa fruticans* was not included in the calculation of this index because of its palm nature, means that no girth at breast height can be measured. The high relative dominance values of the *Avicennia* spp. were responsible for its high importance values. These high importance values (105-190% in Creek A, 167-184% in Creek B, 63-158% in Creek C) were also observed in the French Guiana forests by Fromard et al. (1998) where in mature coastal and adult riverine mangrove sites, *Avicennia* exhibited the highest value of importance values ranging from 144 to 181% followed by the *Rhizophora* species.

The high species diversity in the area may be traced to its geographical location. The Philippines belong to the Indo-Malesia zone of the Indo West Pacific region known to have the highest diversity in terms of major, minor and associated mangrove species (Duke 1992). Chapman (1984) accounted for only 27 species of mangroves in the Philippines, Spalding et al. (1997) reported only 30 species, while Primavera et al. (2004) listed 35 species. These accounts did not include the mangrove fern *Acrostichum* spp. and some mangrove-associates which were included in the listing of Duke et al. (1998). Based on the account of Duke et al. (1998), including the mangrove ferns and the associated species, the Philippines has 40 of the 51 species recorded in the Indo-Malesia zone. Of these 35 species documented by Primavera et al. (2004), 33 can be found in Panay Island and 26 in the present study site. *Bruguiera parviflora* has been added to that list, being documented only during the present mangrove community structure in July 2005 and is first reported in this chapter, making a total of 27 species in the Naisud and Bugtong Bato mangroves. The high diversity of the present study site was also revealed during the conduct of community structure analysis where 10 species were recorded in some of the quadrats sampled (left and right of quadrat 1 in Creek A, quadrat 3 right, quadrat 4 left and quadrat 9 left in Creek C), a single quadrat being 100 m^2 wide. Except for quadrat 9, all these quadrats were located between 500 to 700 m away from the river mouth near the side of the road. Less tree cutting was observed in areas near the roads, maybe for fear of being caught and prosecuted, as compared with the deeper parts of the mangroves where cutting was more prevalent. This may be the reason why the quadrats along the side of the road had the highest diversity. Species diversity obtained in the present work were comparable to the species diversity obtained by Mishra et al. (2005) in

Orissa, India where in the four blocks of mangroves sampled with the number of species ranging from 16 to 24 per block, the species diversity values ranged from 0.72 to 0.82. According to Mishra et al. (2005), riverine mangrove ecosystems are relatively highly diversified, mixed type and richer in species.

Although mangroves are typically distributed from mean sea level to highest spring tide, and perhaps the most conspicuous feature on first glance is the sequential change of tree species parallel to shore, many factors have been suggested to account for the apparent zonation of trees and other associated organisms across the intertidal seascape (Hogarth 1999; Alongi 2002). These include salinity, soil type and chemistry, nutrient content, physiological tolerances, predation and competition (Smith 1992; Semeniuk 1994; Saenger 2002). This sequential changing of species parallel to the shore may be more established in fringing mangroves than in riverine mangroves. Zonation can be a structural feature of mangrove forest in some parts of the world (Smith 1992; Woodroffe 1992). However, unlike open coast habitats where zonation patterns are distinct, mangrove distributions in a riverine environment are extremely variable, particularly if diversity is high (Bunt 1996).

Two types of zonation were described by Saenger (2002), parallel zonation along shorelines and longitudinal zonation along the river. Zonation was not clearly defined in this particular mangrove study site. However, there are some distinct distribution patterns that were observed. For example, the occurrence of *Avicennia alba*, *Avicennia marina* and *Aegiceras corniculatum* towards the mouth of the river was very evident. This may reflect their preference for more frequent tidal inundations. In Iriomote Island, Japan a clear longitudinal zonation along Shiira River was described by Kuraishi et al. (1985) as follows: 0 km *Sonneratia alba*, 0.4 km mixed *Rhizophora stylosa* and *Bruguiera gymnorrhiza*, 0.4-1.5 km pure *Bruguiera gymnorrhiza* and 1.5-2.0 km mixed forest of *Bruguiera gymnorrhiza* and non-mangrove plants. This was also observed in the present work, where *Bruguiera gymnorrhiza* were noticed to grow between 0.5-1.9 km away from the mouth of the river. The higher abundance of *Avicennia alba* from areas not far from the sea was also seen in the schematic diagram of Smith (1992) for the zonation pattern of the mangroves in Peninsular Malaysia where *Avicennia alba* occupies the area near the open sea. The typical *Avicennia marina-Sonneratia alba* association which is very common in some mangroves

(Sasekumar 1974) was not exhibited in this area. The limited extent of distribution of *Avicennia marina* may be due to its tolerance to higher salinity or more frequent inundation, hence it was found closer to the river mouth, together with *Avicennia alba* as compared with *Avicennia officinalis* and *Avicennia rumphiana* which were observed for the most length of the creeks. The same observation in Inhaca Island was reported by Macnae and Kalk (1962) where *Avicennia marina* was commonly found on the seaward boundary. It is the only species that can tolerate desert climate (Macnae 1968) and according to Mandura (1997), it is the only *Avicennia* species known to grow in the Red Sea, although not as trees but as bushes. Stunted growth of *Avicennia marina* in this area was due to high salinity and low precipitation. According to Satyanarayana et al. (2002), in the east coast of India, *Rhizophora mucronata* and *Rhizophora apiculata* were noticed in seaward locations. This is to be expected because the aerial prop roots of *Rhizophora* spp. are more tolerant to longer periods of submergence by flood water (Kathiresan and Bingham 2001; Kathiresan and Rajendran 2005) than any of the aerial roots of the other species, i.e. pneumatophores of *Avicennia* spp. or knee roots of *Bruguiera* spp. However, this was not observed in *Rhizophora* spp. in the present study which were seen some 500 m from the sea. This observation may follow the pattern described by Semeniuk (1994) in species-rich tropical humid areas, where muddy tidal flats may have a sequence from the seaward zone of *Avicennia marina* (trees), *Rhizophora* spp. and *Avicennia marina* (bushes), again. *Heritiera littoralis* and *Bruguiera gymnorrhiza* were being mostly observed on drier land near the edge of the forest. This may reveal the need of these species for lesser tidal exposure. Matthijs et al. (1999) reported the presence of *Heritiera littoralis* and *Bruguiera gymnorrhiza* in less reduced, low-sulphide landward zone. Macnae (1968) described a zone considered as *Nypa* association, occurring landward and upstream of the true mangrove zone, dominated by *Nypa fruticans*, with a few isolated trees, mainly *Heritiera littoralis* and *Excoecaria agallocha*. The presence of the latter two species in the landward zone was observed in the present study but the occurrence of *Nypa fruticans* as a back mangrove in the so-called *Nypa* association zone was not observed, because they were planted everywhere. The present status of this mangrove area shows how zonation can be altered by human activities. The massive planting of *Nypa fruticans* has affected the distribution pattern of mangroves because of the rampant cutting of some species to give way to *Nypa* plantation. The presence of *Nypa fruticans* varying from small

patches to hectares of plantation has altered the zonation pattern which should be apparent if the natural mangrove vegetation in these areas was left untouched.

Topographical profile

Following Cintron and Novelli's (1984) classification, the mangroves of Naisud and Bugtong Bato, Ibajay can be classified as riverine mangroves, a type considered to exhibit the highest level of structural development. This type of mangrove is comprised of a tidal creek and fringing mangrove swamps. The mangrove study site is exposed during low tide and flooded during spring high tide. Most parts of the tidal creeks remain under water at all tides, though some areas may be drained during very low spring tides. Since the whole area is flooded during high tides, crab trapping can be done in the whole area except for *Nypa* plantation areas, which were avoided by the fishers because of the observation that less crab can be caught in these areas. This observation had been confirmed by Langdown (2005).

Creek A was observed to be sloping upstream from the junction where it joins Naisud River to its tip. This ascending slope may be caused by the deposition of sediments coming from pond effluents. A large pond runs parallel along the left side of this creek, covering almost 75% of its length. Increased sedimentation may have caused the creek bed to become shallower. No apparent changes on the height of the bank were noticed throughout the length of the creek. Increased sedimentation is one indirect effect of pond culture (Alongi 2002). Creek beds surrounded by *Nypa* plantations were also observed to be shallower, as seen in Creek B and the tip of Creek C. The deepest part of Creek C corresponds to the area between quadrats 5 and 10 which is a long stretch of natural mangrove forest, not planted with *Nypa* and close to the climax community of *Avicennia* (Fig. 2.2). Mangrove vegetation, especially those species with a complex matrix of roots such as the *Avicennia* spp. and the *Rhizophora* spp. facilitates accretion (Saenger 2002). Most of the sediments carried by the water during ebb tide may have been trapped by the complex root structures in the dense natural mangrove forest on both sides of the creek causing less sediment deposition on the creek bed, thus making this portion of the creek deeper. According to Hogarth (1999), established mangrove forests trap sediment particles and accelerate accretion. It can also be observed that the creek bed towards the mouth of the river

was shallower. The three creeks join Naisud River between quadrats 2 and 3 which are situated along this area. During heavy rains Naisud River carries loads of sediment from the mountains and some of them may have settled near the mouth of the river. According to Ewel et al. (1998), river waters carry heavier sediment load than the ocean tides. Moreover, in this area fresh water from the river mixes with salt water and when this happens, the suspended solids start to flocculate and form larger aggregates whose settling velocity may be an order of magnitude higher than the settling velocity in freshwater (Hogarth 1999). These larger aggregates of fine particles are usually deposited in estuaries (Saenger 2002). Thus, the shallower creek bed on this part of the creek may be caused by heavy sedimentation from river run-off and faster settling rate of heavier flocculants.

In some parts of the creek, the height of the banks was not measured due to the presence of mud lobster mounds. These mounds were very common along both sides of Creek C and the portion of Creek B on the left close to the climax forest of *Avicennia* spp. According to Ashton et al. (2003), these mounds constructed by the mud lobster *Thalassina anomala*, can greatly affect shore topography and hence the composition of the plant community. Presence of lobster mounds may be beneficial to these mangroves since no planted *Nypa* was observed in areas where these mounds flourish.

Environmental parameters and tidal inundation

On a global scale, mangroves are ultimately limited by temperature and precipitation, but at the regional scale the area and biomass of mangrove forests vary in relation to rainfall, tides, waves and rivers (Alongi 2002). Waves, tides, rivers and rainfall affect water circulation by generating turbulence, advective and longitudinal mixing and trapping coastal water, influencing the rate of erosion and deposition of sediments on which mangroves grow. According to Blasco (1984), it has been difficult or impossible to determine with certainty the controlling climatic factors, because primarily these factors are usually interdependent. Regarding zonation and succession of mangrove vegetation, it is generally accepted that they are related to complex local factors among which hydrology and climate are dominant. Among the abiotic ecological factors, soil salinity, soil structure, sea level and tidal inundations are the

main agents controlling the distribution of mangroves (Blasco 1984; Chapman 1984). Salinity of the interstitial soil water has long been recognized as an important factor regulating growth, height, survival, distribution and zonation of mangroves and is regulated by a number of factors such as tidal inundation, soil type, topography, amount and seasonality of rainfall, freshwater discharge from river and evaporation (Saenger 2002). The distribution of mangroves in the present study site may be more affected by tidal inundation rather than salinity. In the absence of rain, salinity in this particular area is rather constant along the creeks and is not affected by the freshwater coming from Naisud River. However, it can significantly drop within hours in the event of a heavy rain which is the main factor that causes salinity variations. This was observed when taking readings from the three creeks. Only small differences in salinity were observed in all quadrats sampled except for Creek B. Creeks A and C had significantly higher salinity than Creek B. Salinity measurements in the first two creeks were taken on sunny days while salinity measurements on Creek B were obtained on a rainy day.

Analysis of the soil composition revealed that all three creeks had silt as the major component of the substrate, followed by sand, then clay. It was observed that silt was highest in Creek A while sand was highest in Creek B. Creek C on the other hand was intermediate between Creeks A and B. The high silt content in Creek A may be caused by the presence of a culture pond right beside it. Alongi (2002) reported that pond culture can increase sedimentation. The effluents coming from the ponds may have caused higher silt composition of the substrate along this creek compared with Creeks B and C. On the other hand, the high proportion of sand in Creek B as compared with the other creeks may be caused by the wide stretch of *Nypa* plantation making the area open, bare and exposed. According to Macnae (1968), in areas where the uppermost intertidal levels are bare of vegetation, the deposits are often sandy. *Nypa fruticans* have no complex root structures capable of trapping sediments. During ebb tide, much of the finest particles are transported with the water back to the river or the sea leaving behind the bigger and heavier particles such as sand to settle on the substrate.

Soil composition may affect distribution of mangroves in a certain area. Chapman (1984) described the requirements of some mangrove species on the west coast of

Malaysia which may explain the distribution pattern of some species observed in the present study site. Based on Chapman's (1984) table of soil requirements for mangrove species, *Avicennia alba* prefers deep mud and brackish water which may explain the limited distribution of the species in the present study. *Avicennia alba* was found only around the vicinity of the area where Creeks A and C join Naisud River. This area is very muddy and has access to freshwater coming from the river. The upper-intertidal location of *Heritiera littoralis* and *Excoecaria agallocha* may be explained by their need for a drier land, flooded only during spring tides. Most of the mangrove species were described to have loam, silt, clay or mud as their substrate of preference. In a study conducted in Gazi Bay, Kenya, less mangrove vegetation was observed in sandy substrate (Fondo and Martens 1998). The significantly lower diversity in Creek B as compared with Creeks A and C may be due to the higher percentage of sand in the substrate which makes it less suitable for settlement to other species of mangroves.

CONCLUSION

Community structure analysis of the mangroves in Naisud and Bugtong Bato, Ibajay, Aklan revealed a highly diverse mangrove area, on the verge of losing its species richness to human activities. The once believed pristine mangrove area has now become a greenbelt to the growing *Nypa* plantation hidden behind the tall *Avicennia* trees. Although no information was gathered in the past to compare with the present findings, based on observations, much has changed for the past decade brought about by the expanding aquaculture pond and *Nypa* plantation. The only remaining untouched area is the climax community of *Avicennia* spp. This may be because, lobster mounds abound in the area, which make it difficult to plant *Nypa* and second, the trees are so large and difficult to cut down and the rough terrain created by the mounds makes it more difficult to bring the logs out of the forest. With almost 30% of the once pure stands of natural mangrove area lost to *Nypa*, the area can still be considered ideal for stock enhancement trials since the habitat is not completely lost. Approximately, 50 ha of the area is inhabited by natural mangroves and is suitable as mud crab habitat. It is beneficial for the wild crabs and ideal for stock enhancement

that most of the mangroves around the vicinity of Creek C where the basin area is located have been left uncut.

No clear zonation pattern was observed in the area except for the greater abundance of *Avicennia alba*, *Avicennia marina* and the *Rhizophora* spp. toward the seaward zone and of *Bruguiera gymnorrhiza*, *Heritiera littoralis* and *Excoecaria agallocha* toward the landward zone. Most of the species are distributed throughout the length of the creeks. This may have been caused by an almost uniform salinity throughout the length of the creek. Salinity varies only in the event of rain. There was no significant difference in pH, soil composition and salinity between creeks which may have caused the lack of clearer zonation pattern of mangroves in the area.

Survey of the topography of the area revealed a single, narrow entrance from the sea. A 1-km basin-like area was also observed in the main creek. This limited access to the sea may limit recruitment of mud crabs in the area which may affect its population and eventually the fisheries that depends on the wild. Having some areas of the mangroves still intact, with the topography limiting recruitment, and the presence of a basin-like area make this particular site suitable for stock enhancement trials of the mud crabs *Scylla* spp. The narrow access to the sea can limit emigration of released crabs hence more crabs will be available within the area to augment decreasing wild populations. With the whole area flooded during spring tide, crab trapping for baseline information gathering and monitoring of released crabs will not be limited to the creeks. This will allow a more uniform and evenly distributed sampling than having traps deployed only along the creeks.

Aside from the relevance of this work to the present study on population, fisheries and stock enhancement trials of the mud crabs *Scylla* spp., the information obtained from the present work may be used as a baseline data for resource management programmes or as a reference in future assessment studies in this particular mangrove ecosystem.

CHAPTER 3

Baseline fisheries assessment

Part of this chapter was presented in the First International Symposium on Mangroves as Fish Habitat, Rosenstiel School of Marine and Atmospheric Science, University of Miami, Miami, Florida, U.S.A., 19-21 April 2006

A manuscript was submitted to *Bulletin of Marine Science* as follows:

Ma. Junemie Hazel L. Lebata, Lewis Le Vay, Jurgenne H. Primavera, Mark E. Walton, and Joseph B. Biñas. Enhancement of fisheries for mud crabs *Scylla* spp. in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines – baseline assessment of species abundance

CHAPTER 4

Estimation of abundance, growth and movement of *Scylla* spp. at the study site

INTRODUCTION

This chapter aims to study the wild population of *Scylla* spp. using a mark-recapture technique employing the use of coded microwire tags and the Jolly-Seber method for analysing open populations. It mainly aims to estimate the abundance, survival and number of new recruits in the wild population based on the recaptures from released tagged crabs. Data from this experiment support the estimates of abundance using CPUE and the number of recruits using monthly mean carapace width obtained from the previous chapter. Growth rates between sexes and size classes are also compared. The general movement of the crabs from the release site to the collection site is also reported.

Ecologists and wildlife biologists have long used the technique of mark-recapture, in which animals are captured and then physically marked or tagged in some way with a personal identification number or code before being released (Gibbons and Andrews 2004). This widely used method for obtaining information about populations employing the use of tags or marks either externally or internally attached to the animal is also known as capture-recapture or tag-recapture (Schwarz and Seber 1999). The use of these methods to estimate the size of animal and even human populations dates back more than two centuries ago. Aside from population estimates, capture-recapture studies also provide quantitative estimates of growth, movement, and mortality (Seber 1982). They have long been employed in ecological and fisheries research to which they are particularly suited because direct observation of an aquatic population is often difficult, and commercial and experimental fishing are well-suited to capture samples of particular organisms (Bell et al. 2003). The fact that some animals must be recovered means that the methods have mostly been applied to commercial species where fishing effort allows an acceptable recapture rate (Freire and Gonzalez-Gurriaran 1998).

Tags may either be external or internal. External tags vary from simple external markings such as paints (Bell et al. 2003) or coded plastic tags glued externally to the animal (Brousseau et al. 2002) or the mutilation of the external part of the organism such as clipping fins or spines (Robertson 1996) or making notches on the carapace

(Cagle 1939). In early 1960s, Jefferts et al. (1963) introduced an internally implanted, biologically inert, coded microwire tag that has become commonly used in finfish research. Since then, tags have evolved from this simple coded microwire tag or CWT (van Montfrans et al. 1991; Northwest Marine Technology 2000; Sharp et al. 2000; Jones and Coulson 2006) which were also used in the present study, to more sophisticated ones like the passive integrated transponder (PIT) (Gibbons and Andrews 2004) or ultrasonic tags (Gonzalez-Gurriaran et al. 2002; Stone and O'Clair 2002). Recent developments in technology have created new types of tags and increased the range of data that can be collected. The simple mark-recapture markers used for population studies evolved in recent years into studies using multi-purpose acoustic tags (Senkowsky 2003). As a tool to monitor migration patterns or movements which allows tracking of the animal bearing the tag, electronic data storage tags with ultrasonic transmitters are now available (Gonzalez-Gurriaran et al. 2002; Maitland et al. 2002; Stone and O'Clair 2002). Radio tagging is commonly used not just to estimate survival and abundance, but also to analyse habitat, home ranges and movement of animals (Schwarz and Seber 1999). The different tagging methods used for fish, crustaceans and molluscs, including their advantages and disadvantages are discussed in Thorsteinsson (2002).

In crustacean research, both external and internal tags have been used. External tags may include simple physical markings like uropod or tail-fin clipping or eye-stalk rings in shrimps; dart, T-bar (Bell et al. 2005) and sphyrion spaghetti tags in lobsters (Campbell 1983), or T-bar (Moser et al. 2002; Pillans et al. 2005) and Floy anchor tags (Hill 1975; Williams and Hill 1982) in crabs. Diaz and Conde (1989) also used small, sequentially numbered, auto-adhesive paper affixed onto the carapace of the crabs. The use of these types of tags for long term experiments in crustaceans, however, presents problems as they tend to be lost with the exoskeleton during moulting (Sharp et al. 2000). Despite this problem, tagging of crabs dates back to as early as 1920s (cited by Cronin 1949), where external tags made of plastic or wire with serial numbers were attached to large crabs to study patterns of migration in adults.

Internal tags used in crustacean include coded-wire and florescent elastomer tags in shrimps (Bell et al. 2005); coded microwire (Uglem and Grimsen 1995; Sharp et al.

2000), florescent elastomer (Uglem et al. 1996) and internal anchor tags (Melville-Smith and Yuk 2002) in lobsters; and coded microwire tags and injection of coloured plastic polymer (Davis et al. 2004) in crabs; with CWT being the most widely used in *Scylla* spp. (Le Vay et al. 1999; Ut 2002; Walton 2006), the blue crab *Callinectes sapidus* (van Montfrans et al. 1986; van Montfrans et al. 1991; Davis et al. 2004; Eggleston et al. 2004) and the swimming crab *Portunus tuberculatus* (Okamoto 1999; Okamoto 2004). Lately, natural tags such as the stable isotopes and genetic tags have also been used in shrimps (Brown et al. 2003; Bell et al. 2005).

Aside from population studies, the use of marking or tagging to track movement of crustaceans has also been employed by Hill (1978), Hyland et al. (1984) and Gonzalez-Gurriaran et al. (2002) for crabs; and Melville-Smith and Yuk (2002) and Mills et al. (2004) for lobsters.

Estimating the size of wild populations plays an important role in managing harvested populations and conserving rare and endangered species. An improved understanding of growth rates is also a crucial part of stock assessments that are length based (Lee et al. 2006). One of the most common techniques to obtain such information has been to capture, mark, release and later recapture of individuals (Miller et al. 2005). In most cases, tagging presents different problems related to handling the animal, induced mortality or tag shedding. However, despite these limitations, mark-recapture methods have been widely used in crustaceans because they provide data of great interest in population dynamics and large scale migration patterns (Freire and Gonzalez-Gurriaran 1998). Mark-recapture studies have been used for shrimps (Iversen 1962; West and Chew 1968), lobsters (Morgan 1974; Sharp et al. 2000), crayfish (Jones and Coulson 2006), and crabs (Hill 1975; Bennett 1979; Williams and Hill 1982; Diaz and Conde 1989; van Montfrans et al. 1991; Moser et al. 2002; Bell et al. 2003). Several studies have been conducted with the mud crab *Scylla* (Hill 1975; Williams and Hill 1982; Robertson and Piper 1991; Le Vay et al. 1999; Moser et al. 2002; Ut 2002). The present study is the first of its kind to be done in natural mangroves in the Philippines, though Walton (2006) undertook a simultaneous study in replanted mangroves in Kalibo, Aklan.

Choosing a method appropriate for estimating animal population parameters such as population size or survival rate mainly depends on the nature of the population investigated (Schwarz and Seber 1999). Literature relating to the estimation of these parameters continues to grow rapidly. However, most of them are just elaborations from the simple most basic method called the Petersen method or the Lincoln index (Seber 1982; Greenwood 1998; Bell et al. 2003). This method assumes that the population is closed, wherein no gains (births or immigration) or losses (deaths or emigration) occur during the course of the study and only one session of catching and marking and one session of recapture take place (Greenwood 1998). The following are the Petersen assumptions for mark-recapture methods for closed populations (Seber 1982):

1) the population is closed, so that the population estimate N is constant;
2) all animals have the same probability of being caught in the first sample;
3) marking does not affect the catchability of an animal;
4) the second sample is a simple random sample, such that each of the possible samples has an equal chance of being caught;
5) animals do no lose their marks in the time between the two samples; and
6) all marks are reported on recovery.

A simple extension of the Petersen method is the Schnabel method, which follows the same assumptions as above but is appropriate to situations where animals are captured on several occasions, such that all unmarked animals in each capture are marked before being released (Seber 1982; Greenwood 1998). Another method used for closed populations is the Burnham and Overton method, which assumes that the probability of recapture differs between marked animals. This assumption, according to Greenwood (1998), is perhaps the most common cause of bias in capture-recapture studies.

The first general model for open populations was developed independently by Jolly and by Seber (both in 1965). This method requires that the number of unmarked animals be recorded at each sampling occasion and that they should be marked and returned to the population in order to establish capture histories of marked animals

(Schwarz and Seber 1999). The following are the Jolly-Seber assumptions for mark-recapture methods for open populations (Seber 1982):

1) marked animals should be evenly distributed in the population such that every animal in the population, whether marked or unmarked, has the same probability of being caught in the *i*th sample, given that it is alive and in the population when the sample is taken;
2) every marked animal has the same probability of surviving from the *i*th to the (*i*+1)th sample and of being in the population at the time of the *i*+1 sample, given that it is alive and in the population after the *i*th release;
3) every animal caught in the *i*th sample has the same probability of being returned to the population;
4) marked animals do not lose their marks and all marks are reported on recovery; and
5) all samples are instantaneous, i.e. sampling time is negligible, and each release is made immediately after the sample.

The mud crab population in the present study is considered open, defined as one that changes through addition by 'birth', recruitment or immigration or removal by death (natural, fishing mortalities) or emigration (Seber 1982; King 1995; Pollock 2000). It has often been assumed that populations of marine species with planktonic larvae are demographically "open" (Bilodeau et al. 2005). Hence, this study used the Jolly-Seber method for analysing open populations in Greenwood (1998). All, but one of these assumptions, were met in this present work. Marked animals recaptured were not returned because they were dissected to remove tags for reading, thus missing assumption no. 3. This special case of the general capture-recapture model, where animals are recaptured only once, is sometimes called tag-recovery (Schwarz and Seber 1999), and is a recognized modified Jolly-Seber method. Assumption number 3 is replaced by a new assumption which states that "every tagged animal caught or dying in the *i*th sample has the same probability of being found and reported at time *i*" (Seber 1982). According to Seber (1982), although loss on recapture in tag-recovery studies is 100%, this represents no loss of information, for once the released animal is recaptured, it has yielded all information between time of release and recapture which include the probability of being recapture and the probability of surviving in the

population. This method is commonly applied for analysing recaptures obtained from commercial catches and is best suited for exploited populations such as fisheries (Seber 1982). This makes this method appropriate for the present study on mud crab populations. With the baseline information on the population and fisheries of *Scylla* spp. obtained from crab landings for the previous 21 months (April 2002-December 2003, see Chapter 3), the present study was conducted to make an assessment of the abundance of wild *Scylla* spp. in the natural mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Panay Island, Philippines. The result of this work and that of the population and fisheries studies in Chapter 3 will be used to assess the potential for stock enhancement of the population through release of hatchery-reared crabs.

MATERIALS AND METHODS

Study Site

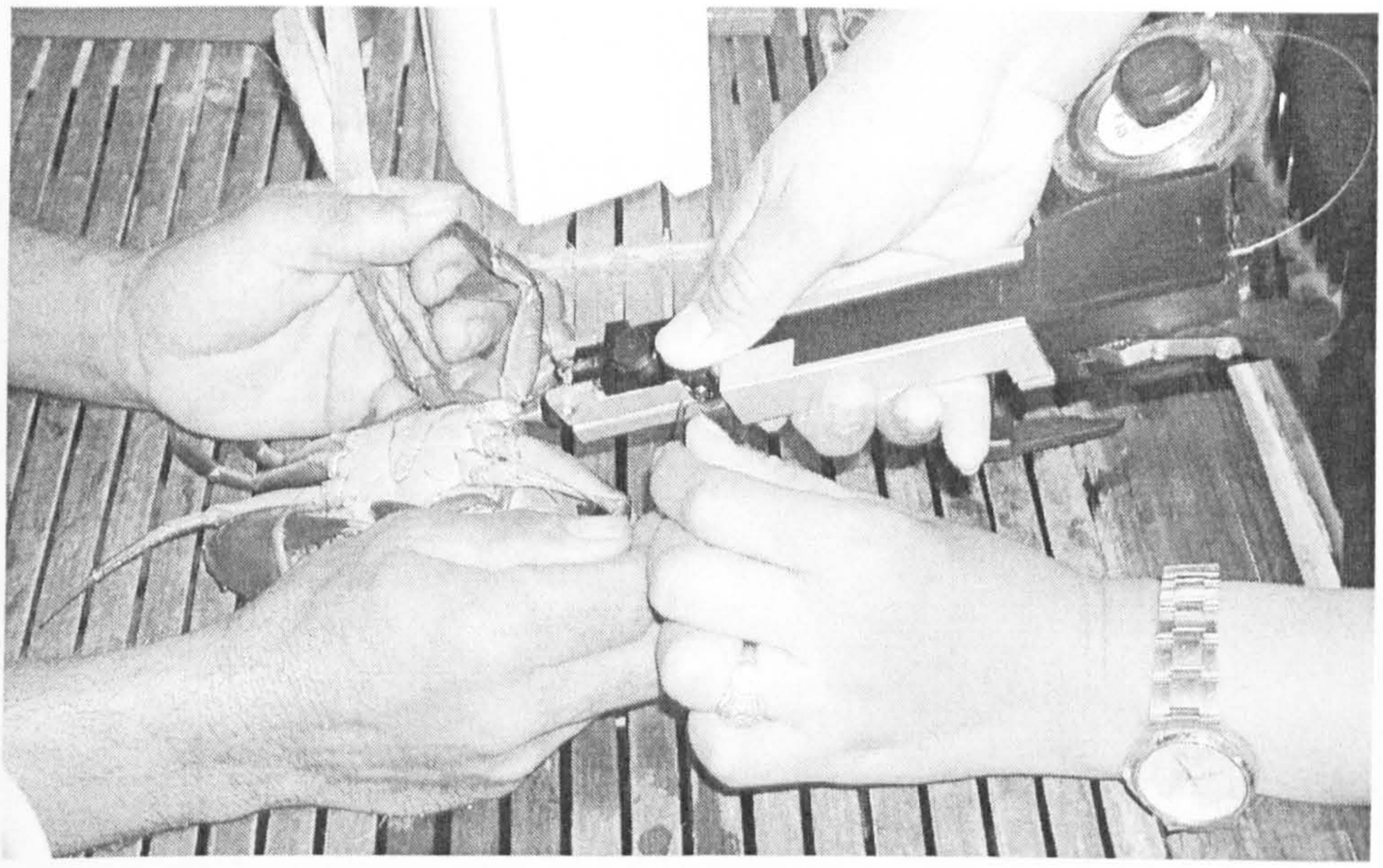
The study was conducted in the same area as population and fisheries monitoring (Chapter 3) and which was described in detail in Chapter 2. The Naisud River and its tributaries were divided into four release sites and the mangrove area surrounding it into six collection areas (Fig. 4.1). Release sites were identified using a hand-held Garmin GPS 12 XL for easy access in the succeeding release sessions. Release of marked crabs was conducted from February to April 2004. Every spring tide (twice a month), daily crab catches were tagged and released such that catches in day 1 were released in Site 1 and so on until the 4^{th} day to complete release of marked crabs in all 4 sites. Crab collection in each area adjacent to the release sites was delayed for a day immediately following the release of marked crabs, in order to allow even dispersal and evade immediate recapture. Each of the six collectors was assigned a specific collection area shown in Figure 4.1 to spread sampling stations evenly within the study site to approximate uniform sampling and equal probability of capture for both tagged and untagged crabs. This also allows description of at least the general movement of crabs within the mangroves from the release site to the collection area during the period of the experiment.

Figure 4.1. Release (top) and collection sites (bottom) of marked *Scylla* spp. in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; February 2004-February 2005. White and black arrows with numbers point to the part of creek where releases started and ended, respectively.

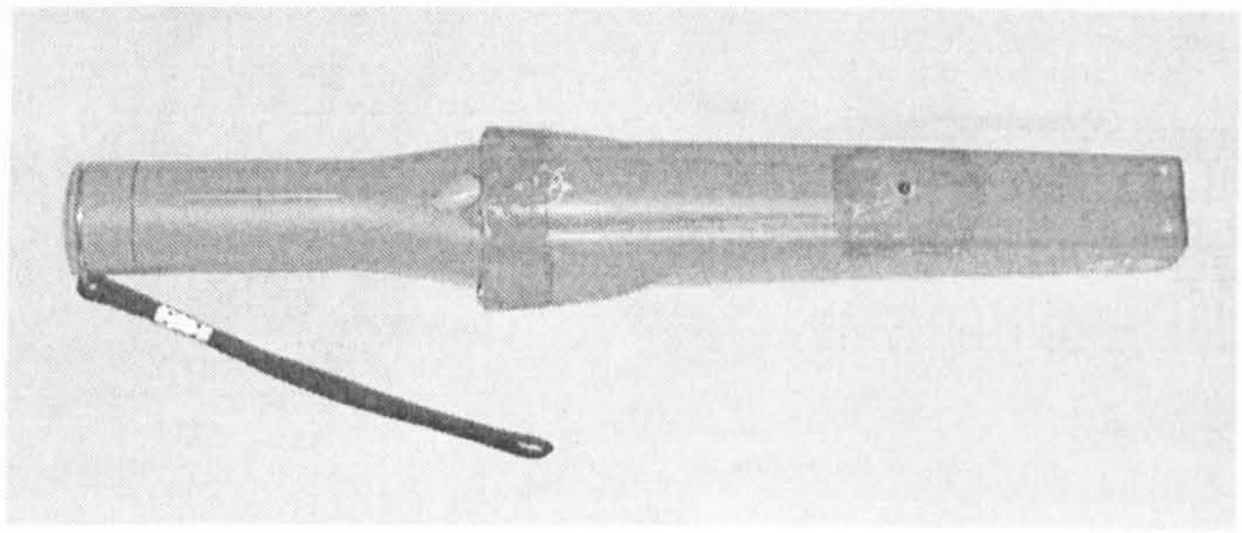
Tagging and release of crabs

All crabs for this experiment were caught using bamboo traps only. Spring tide catches of *Scylla* spp. during the duration of the experiment were purchased for tagging and release. Prior to tagging, crabs were identified to species level following Carpenter and Niem (1998) and Keenan et al. (1998). Carapace width (mm CW) and weight (g BW) of crabs were individually measured using a dial calliper and a balance, respectively. Sex and ovarian maturation of female crabs were noted and categorized as immature, mature, gravid, berried and spent using the ovaries and abdomen morphology as an indicator of maturity (Overton and Macintosh 2002). Each crab was then injected at the base of its third walking leg on the right (Fig. 4.2a) with a magnetic decimal sequential microwire tag (1.1 mm long x 0.25 mm dia), commonly called a coded wire tag (CWT), using a hand-held multi-shot microwire tag injector (Northwest Marine Technology 2000). Successful tag insertion was checked using a hand-held wand metal detector (Fig. 4.2b) and each marked crab immediately placed in a container with seawater to encourage clotting of haemolymph over the insertion point, thus preventing tag loss (Ut 2002). Alternately, after tagging each crab, a reference tag was taken and taped opposite the information (species, sex, CW, BW) referring to the marked crab. The sequential tags are designed for use where identification of small batches or individual specimens is desired and saving of a reference tag (preceding in the sequence) allows identification of individual samples. When all crabs had been tagged, they were placed in a bag made from dried palm leaves and transported to the release site. Crabs were released in the water one at a time (Fig. 4.3) and distributed at an even distance from the start to the end of the release site. This procedure was repeated for four days to complete distribution of crabs in all four sites. Table 4.1 shows the data summary for each session; the dates of release, release sites, number of crabs released per site and the number of crabs recaptured.

Temperature and salinity were measured using a hand-held YSI conductivity meter. Measurements were taken at the release sites when crabs were released, during the commencement of high tide. Temperature ranged from 26.11 to 30.43°C (mean ± S.E.=28.03 ± 0.38) and salinity from 6.8 to 34.2 p.s.u. (mean ± S.E.=22.62 ± 2.57).



a)



b)

Figure 4.2. a) Hand-held multi-shot microwire tag injector used to insert the magnetic decimal sequential microwire tag (1.1 mm long x 0.25 mm dia) at the base of the third walking leg on the right of the crab; and b) hand-held wand metal detector used to check for successful tag insertion during tagging and for recaptures during monitoring of crab catches.

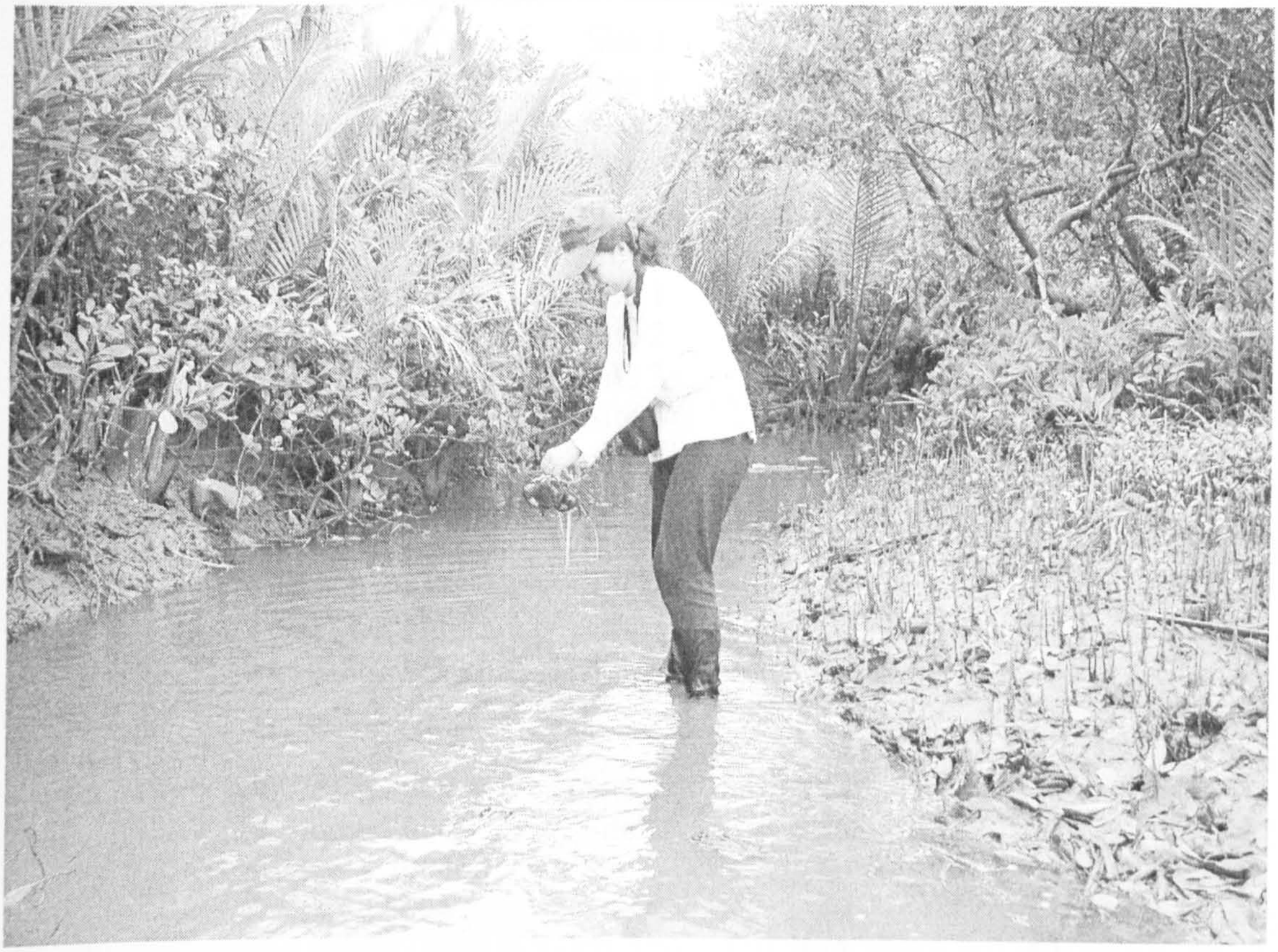


Figure 4.3. Releasing crab individually in the water for mark-recapture study of *Scylla* spp. in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; February-April 2004.

Table 4.1. Data summary of *Scylla* spp. captured, tagged and released in each site for every session (February 2004-April 2004) and the total number and percent recaptured (February 2004-February 2005) from the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines.

Session	Date captured tagged and released	Release site	Number released	Number recaptured	%
1	5 February 2004	Site 1	38	9	23.68
	6 February 2004	Site 2	26	9	34.62
	7 February 2004	Site 3	10	6	60.00
	8 February 2004	Site 4	21	4	19.05
Total			**95**	**28**	
2	19 February 2004	Site 1	55	13	23.64
	20 February 2004	Site 2	41	13	31.71
	21 February 2004	Site 3	39	23	58.97
	22 February 2004	Site 4	36	16	44.44
Total			**171**	**65**	
3	5 March 2004	Site 1	38	12	31.58
	6 March 2004	Site 2	31	7	22.58
	7 March 2004	Site 3	26	15	57.69
	8 March 2004	Site 4	31	18	56.25
Total			**126**	**52**	
4	18 March 2004	Site 1	48	14	29.17
	19 March 2004	Site 2	25	6	24.00
	20 March 2004	Site 3	40	25	62.50
	21 March 2004	Site 4	30	7	23.33
Total			**143**	**52**	
5	11 April 2004	Site 1	72	17	23.61
	12 April 2004	Site 2	53	19	35.85
	13 April 2004	Site 3	15	13	86.67
	14 April 2004	Site 4	18	8	44.44
Total			**158**	**57**	
6	20 April 2004	Site 1	55	20	36.36
	21 April 2004	Site 2	22	13	59.09
	22 April 2004	Site 3	24	14	58.33
	23 April 2004	Site 4	28	13	46.43
Total			**129**	**60**	

Highest temperature and salinity were both observed in May 2004 while lowest in January 2005 and August 2004, respectively.

Monitoring of recaptures

All *Scylla* spp. caught on the spring tide following the release session were purchased and checked for the presence of marked samples. Unmarked crabs were tagged for release on that spring tide session while marked ones were kept for processing. Tagging of unmarked crabs from spring tide catches continued until the last release on the 6th session, the 2nd spring tide of April 2004. Monitoring of recaptures was done starting from the 2nd session, the 2nd spring tide of February 2004 until the 27th session, the 2nd spring tide of February 2005.

Figure 4.4 shows the processing of recaptured marked crabs. First, they were segregated by source or collection area. Then, individually, the species was identified, sexual maturation determined and CW and BW measured. The third walking leg on the right of each marked crab was then removed and the tag dissected from its muscle tissue. The tags were not read immediately after dissecting from each of the crabs but instead, they were taped into a log book opposite the information (collection site, species, sex, CW, BW) referring to the marked crab. Tags were read after the end of each sampling session.

Reading of tags

Before reading the tags, both from the recaptured marked crabs or the reference tags, they were soaked in tap water added with few drops of Zonrox™ bleaching solution (5.25% by weight sodium hypochlorite) to easily remove sticky muscle tissue or adhesive material from the tape and fibres from the paper where they had been taped. Soaking was done in a container with small shallow wells to keep each individual tags away from each other. When doing this, care must be taken to keep the magnetized tags from attracting each other. Each tag was then removed from the soaking solution, rubbed between the thumb and the forefinger to remove any sticking materials, attached to the holder and read using an Olympus SMZ 10 dissecting microscope at 10x binocular micrometer lens x 4x objective lens (Fig. 4.5). Codes were recorded for

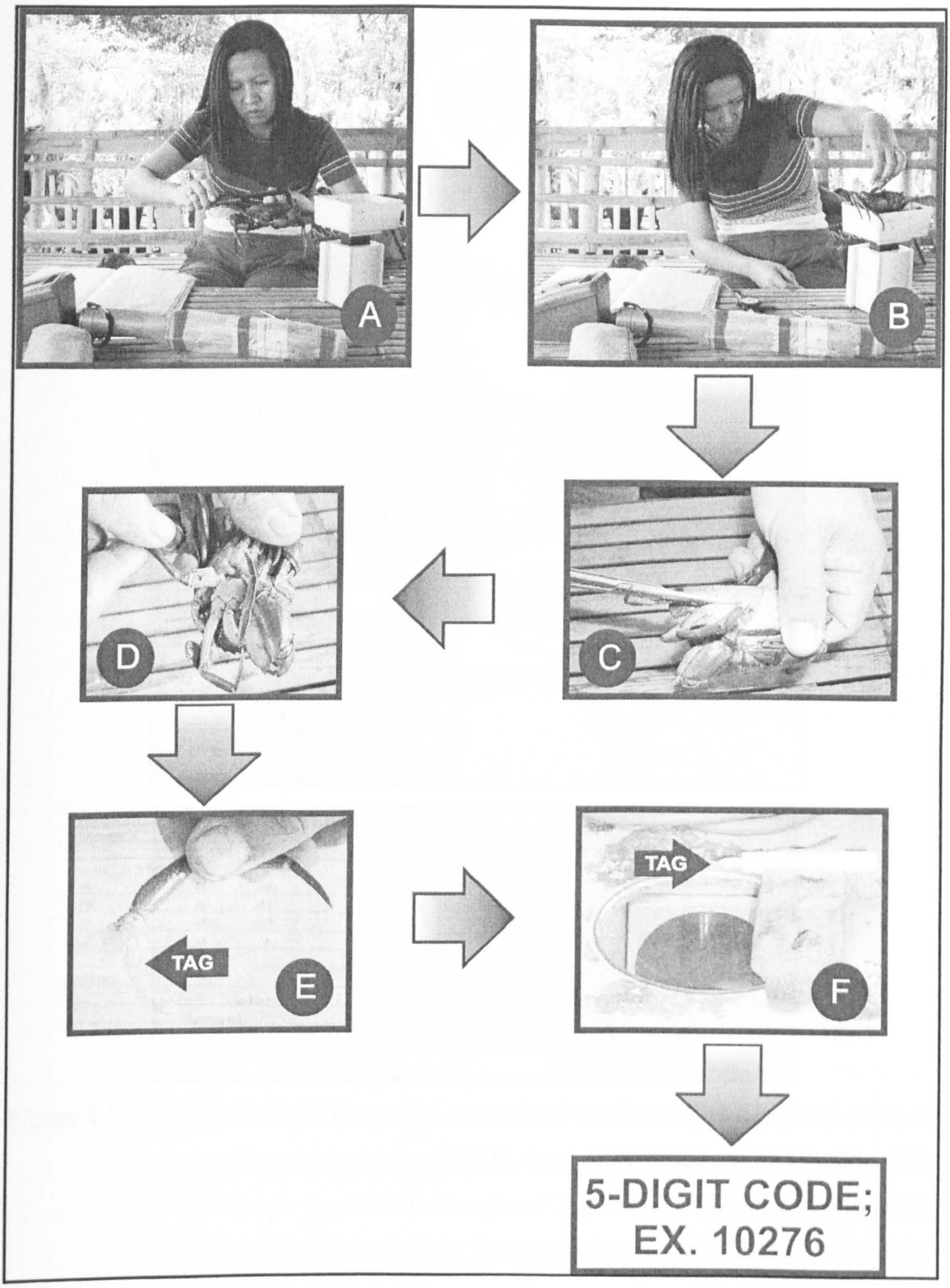


Figure 4.4. Processing of recaptured marked crabs. Each recaptured *Scylla* spp. was individually measured for carapace width and body weight (A and B), its third right walking leg cut and removed (C and D), the coded microwire tag dissected from its leg muscle tissue (E), and attached to the tag holder (F) for reading under the dissecting microscope to get the 5-digit code.

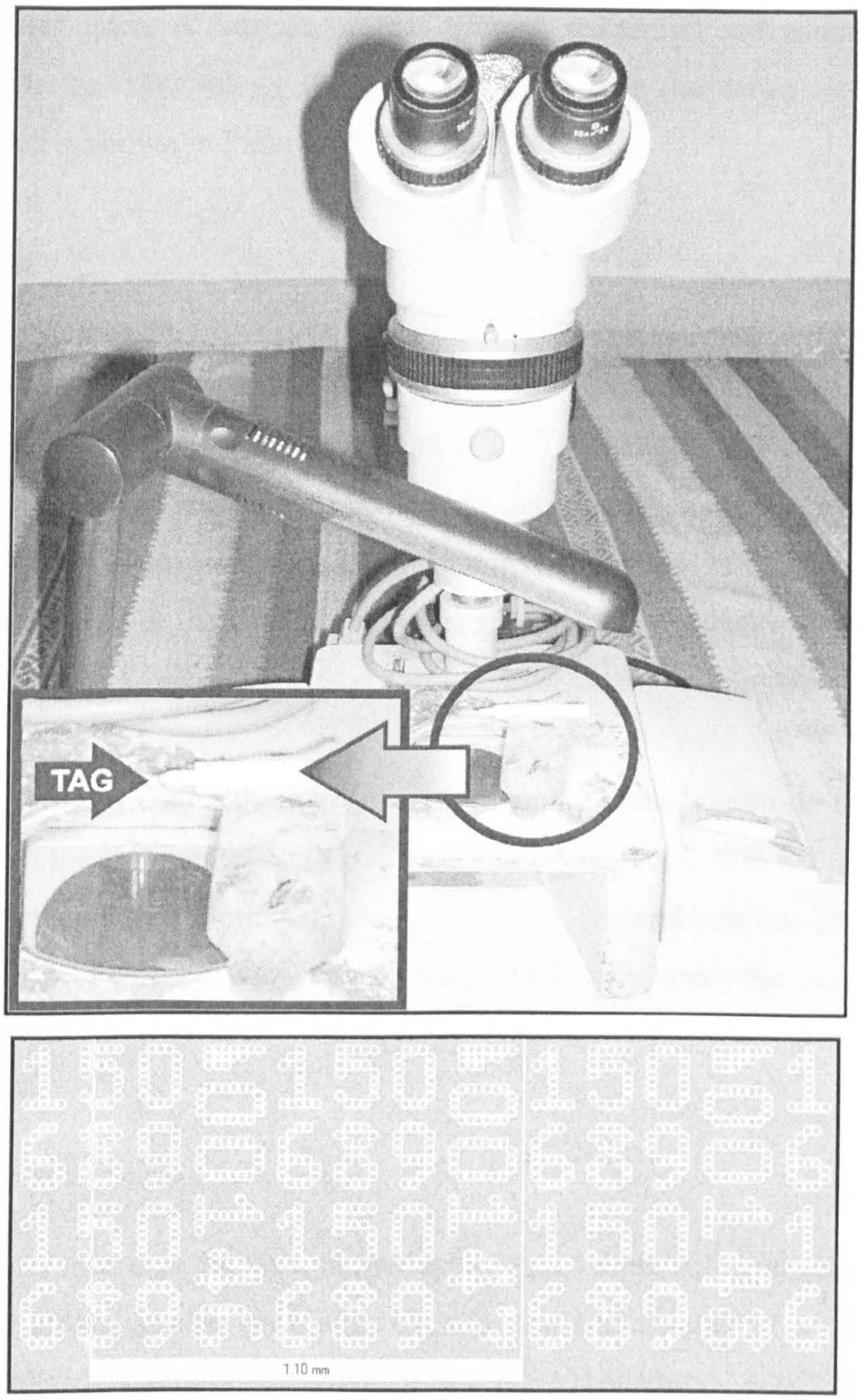


Figure 4.5. Olympus SMZ 10 dissecting scope (top) used to read the decimal codes of the magnetic microwire tag (CWT). Inset shows the enlarged image of the tag attached to the holder. Sample of a decimal sequential tag (bottom) adapted from the manual of the Northwest Marine Technology (2000). The numbers running repeatedly across the circumference of the tag represent the agency (16), data 1 (58) and data 2 (09). The sequence (00146) is shown running parallel around the tag's circumference. The small flag marks the start of the sequence and the agency, data 1 and 2 numbers. The white line shows the length of the tag.

use in analysis later. A sample of the decimal sequential tag adapted from the Northwest Marine Technology (2000) manual showing the sequence of the code being recorded is shown in Figure 4.5.

Data analysis

Data were analysed from February 2004, the commencement of the release session for the mark-recapture experiment, to February 2005 when the last marked crab was recaptured. A total of 352 tagged crabs were recaptured, however, only 314 were analysed, since 38 of the samples were recaptured within the release session (3 days maximum interval). For the analysis, all three species were pooled as *Scylla* spp. since there were only one *S. serrata* and two *S. tranquebarica* in the samples.

During the period covering the mark-recapture experiment, beginning from the start of release until the last marked crab was recaptured, data collection for the population and fisheries studies was continued. Monthly species, sex and size compositions, yield and CPUE from February 2004 to February 2005 were analysed and reported in Chapter 3.

Growth

The use of reference tags between individually tagged crabs allowed identification of each released animal during recapture. Growth rate was calculated using regression analysis of individual growth in length (mm CW at recapture – mm CW at release) and in weight (g BW at recapture – g BW at release), each plotted against time or days (interval between release and recapture) following this general linear regression model:

$$y = a + bx; \qquad \text{(Equation 1)}$$

where: y = dependent variable; final CW or BW – initial CW or BW in mm or g

a = intercept; value of y at $x = 0$

b = slope of the line; the change in y for every unit change in x

x = independent variable; day of recapture – day of release in days

Regression analysis was first done to all crabs recaptured, regardless of their size and sex, to get the growth rate of *Scylla* spp. as a whole, in terms of length (mm CW d^{-1}) and weight (g BW d^{-1}). Then, they were grouped into size classes by length and by weight, and into sexes. Crabs were classified into 7 length size classes at 10 mm CW interval (<41, 41-50, 51-60, 61-70, 71-80, 81-90, >91 mm); 4 weight size classes at 50 g BW interval (<50, 50-99, 100-149, >150 g); and into sexes (male and female). Growth rates were compared between the 7 length size classes, 4 weight size classes, between sexes by length and between sexes by weight to check for differences in growth rates within each group. All analysis and graphs were done using Minitab version 14.

The Gulland-Holt plot was used to estimate K, L_∞ and W_∞. The relationship defined by a straight line when individual growth rates (Equation 2) were plotted against the mean size between release and recapture (Equation 3) following a general linear regression model (Equation 1) is called the Gulland-Holt plot (King 1995).

$$\text{Growth rate} = \frac{L_2 - L_1}{t_2 - t_1} \qquad \text{(Equation 2)}$$

$$\text{Mean length} = \frac{L_2 + L_1}{2} \qquad \text{(Equation 3)}$$

where: L_1 = initial CW or CW at time of release

L_2 = final CW or CW at time of recapture

t_1 = date of release

t_2 = date of recapture

From the general linear regression model, $y = a + bx$, the growth parameters size infinity (L_∞ and W_∞) and K were estimated, such that:

$$K = -b \qquad \text{(Equation 4)}$$

$$L_{\infty} = \frac{-a}{b} \qquad \text{(Equation 5)}$$

where: K = Brody growth coefficient; the rate of growth to reach asymptotic length

L_{∞} = asymptotic length; the length at which growth rate is theoretically zero

Estimates obtained using the Gulland-Holt plot were then compared with the estimates obtained using Beverton's approach to estimating L_{∞} and W_{∞} such that the maximum size occurring in a well sampled stock is divided by a constant (Pellegrin et al. 2001; Lee et al. 2006), as follows:

$$L_{\infty} = \frac{L_{max}}{0.95} \qquad \text{(Equation 6)}$$

where: L_{max} = maximum length occurring in a well sampled stock in a given area

0.95 = constant

This is based on the observation that, in general, the oldest individuals of a stock grow to reach about 95 percent of their asymptotic size (Pellegrin et al. 2001). Note that Equations 2-6 were also applied to weight data.

Population estimate, mortality and survival

The Jolly-Seber method for estimating population parameters was used in analysing mark-recapture data in the present work. According to Greenwood (1998), Jolly-Seber is the method of choice for open populations with individually marked animals. The Jolly-Seber method which requires that the number of unmarked animals be recorded at each sampling occasion and be marked and returned to the population because estimation of survival relies mainly on the capture histories of marked animals (Schwarz and Seber 1999) makes it an appropriate method for analysing the present mark-recapture data. Moreover, most models for tag-recovery (recapture with removal), emphasize estimating survival rather than abundance and only the Jolly-

Seber model can estimate both abundance and survival (Schwarz and Seber 1999). The Jolly-Seber model was also used by Fitz and Wiegert (1992) in analysing population dynamics of the blue crab utilizing the recapture with removal method. Analysis of mud crab data was carried out using the Simply Tagging 1.31 software (Pisces Conservation 1999), a program designed for calculating population size from mark-recapture studies using the Jolly's method equation:

$$\bar{N}_i = \frac{\bar{M}_i n_i}{r_i}; \qquad \text{(Equation 7)}$$

where: $\bar{N}_i$ = estimate of the population on day i

$\bar{M}_i$ = estimate of the total number of marked animals in the population on day i

n_i = total number of animals captured on day i

r_i = total number of marked animals recaptured on day i

From the population estimate, crab density was then calculated based on 50-ha mangrove area. Although the whole mangrove area approximately covers 70 ha, the areas designated for collection for this specific experiment exclude an area around the middle creek where no fisherman fished because of its sandy-rocky nature and the area on the west side near human settlements (see Chapter 2). Mortality was then obtained from the survival values (Φ) generated by the Jolly-Seber analysis as a difference between 1 and the survival estimate ($1 - \Phi$).

Relative abundance in terms of catch per unit effort (CPUE) expressed as the number of crabs $gear^{-1}$ d^{-1} was also analysed per session to complement the Jolly-Seber analysis. Mean carapace width (mm CW) was also analysed per session as a possible determinant of recruitment. The full analyses of monthly CPUE and mean CW from April 2002 to November 2005 are reported in Chapter 3.

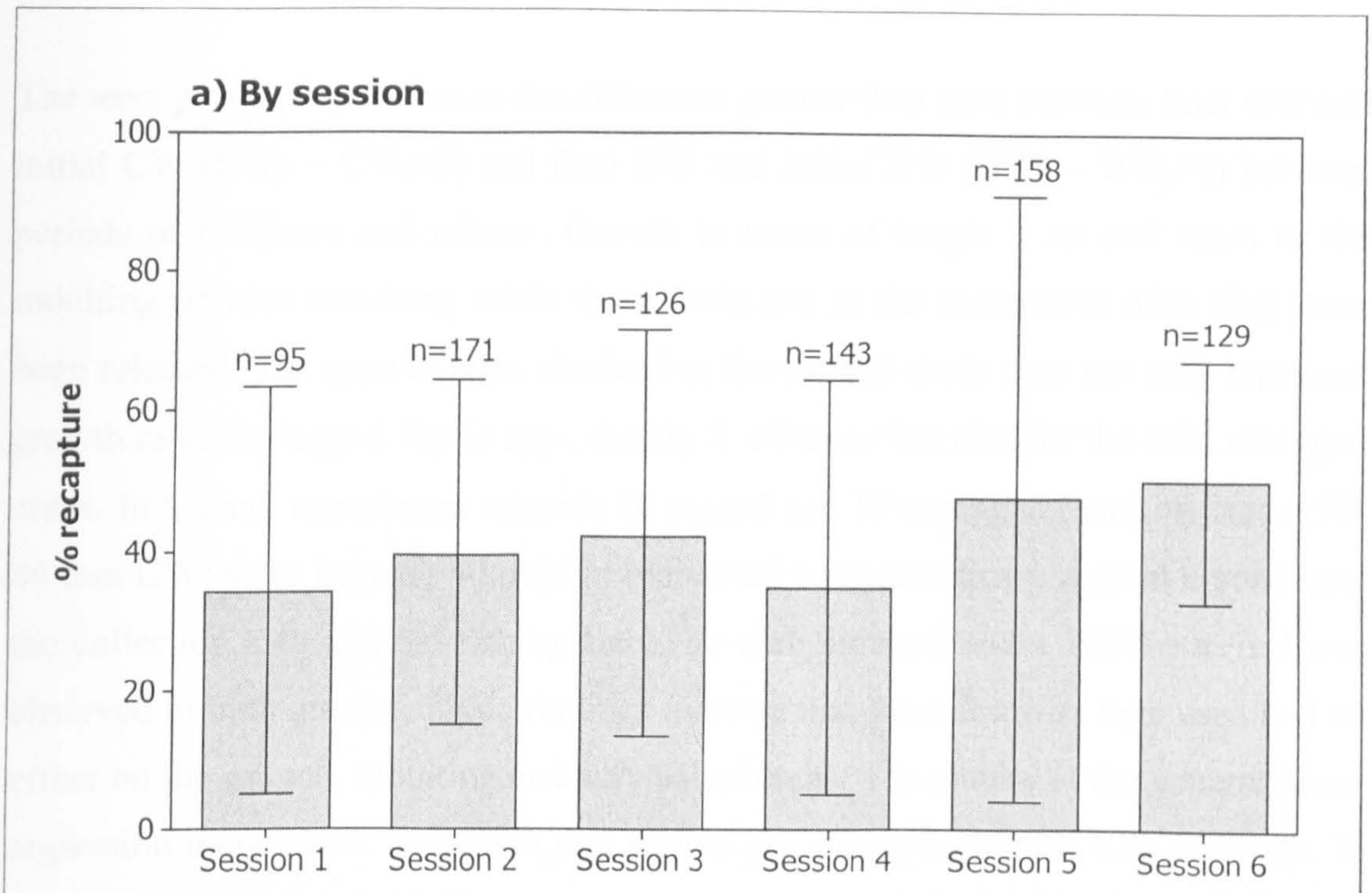
Movement

One of the limitations of this study was not being able to quantify crab movement in terms of distance travelled from the day of release to the day of recapture since crabs were not released at one point only but were evenly scattered over the creek and no tracking device like the simple one utilized by Hill (1978) or the complex one used by Maitland et al. (2002) or Gonzalez-Gurriaran et al. (2002) was used in this experiment. However, general displacement from one site to another and the duration of these movements were noted and presented in the results.

RESULTS

Of the 822 crabs captured, tagged and released from the first spring tide of February 2004 to the second spring tide of April 2004 (6 sessions), 98.5% were *Scylla olivacea*, and only 0.7% each *S. serrata* and *S. tranquebarica*. Of the 314 crabs recaptured from the second spring tide of February 2004 to the first spring tide of February 2005 (25 sessions), 99.0% were *S. olivacea*, 0.3% *S. serrata* and 0.6% *S. tranquebarica*. Because almost all crabs recaptured were *S. olivacea*, it was decided to refer to the recaptures as *Scylla* spp., to include one *S. serrata* and two *S. tranquebarica* from the samples. There was an almost 1:1 male to female ratio among the crabs released, 49.6% females and 50.4% males, consistent with the sexual ratio of crabs in Chapter 3 Female crabs were highly dominated by immature ones with 96.3%, while only 0.7% were mature, 2.7% gravid and 0.3% spent ones.

Using two-way ANOVA, no significant differences were observed in the mean percent recaptures between sessions ($p>0.05$), the highest recapture of 50% was obtained from the crabs released in session 6 and the lowest recapture of 34.4% from session 1. However, there were significant differences in the mean percent recapture between sites ($p<0.001$), with recaptures from site 3 (64.0%) significantly higher than from sites 1 (28.0%) and 2 (34.6%) but not from site 4 (39.3%) which in turn had no significant difference from sites 1 and 2 (Fig. 4.6).



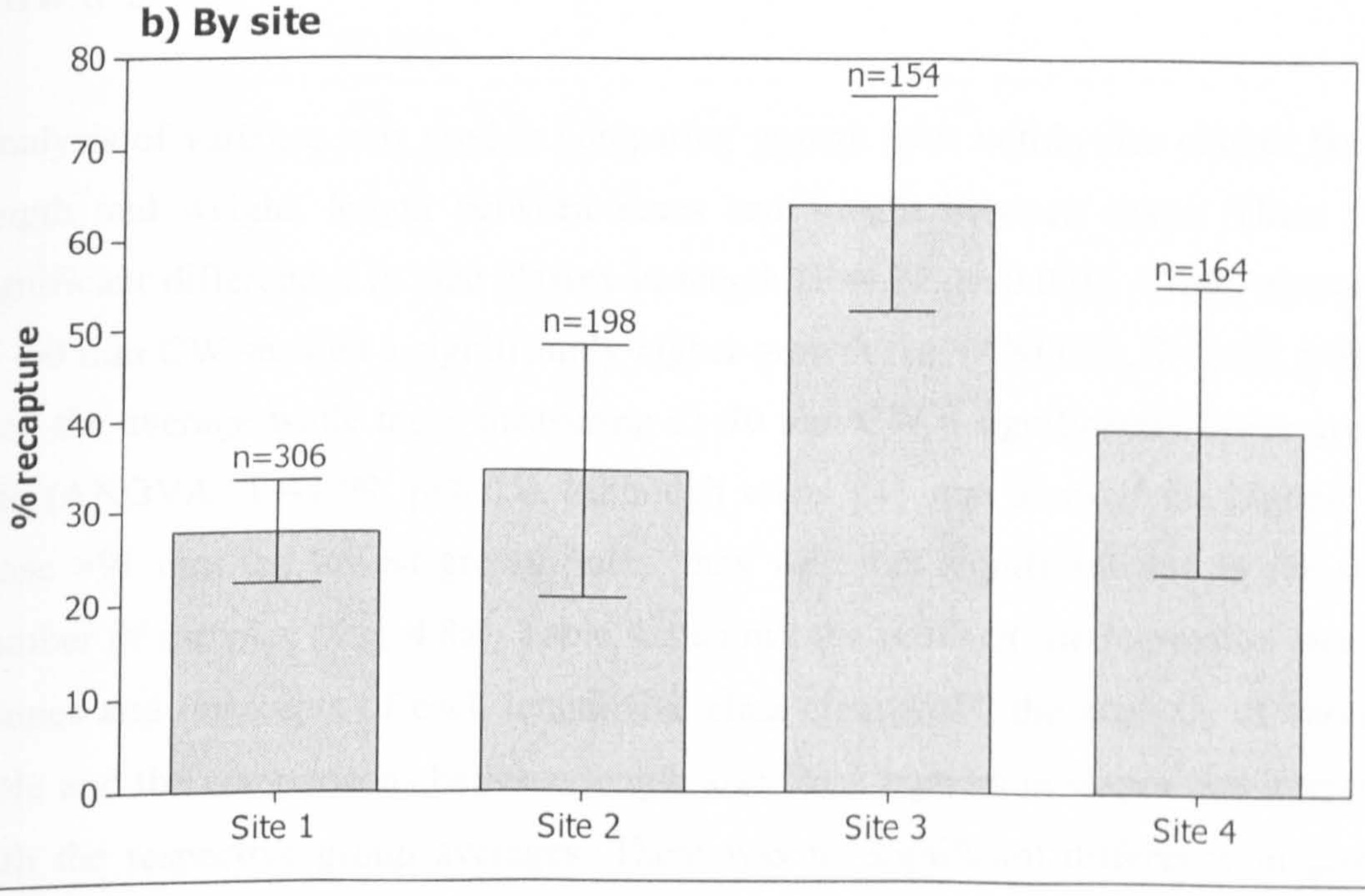


Figure 4.6. Mean percentage recovery by release a) session and b) site of *Scylla* spp. released in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, February 2004-February 2005. Error bars represent 95% confidence interval from the mean; n, the number of crabs released per session or site.

Growth

The term growth here refers to the difference greater than zero between final CW and initial CW ($CW_F - CW_I>0$) and final BW and initial BW ($BW_F - BW_I>0$) between periods of recapture and release. Growth in terms of length is an end result of the moulting process occurring while these crabs are in the mangroves after they have been released. The growth rates obtained in the present study may not only represent growth rates for tagged *Scylla* spp., mainly *S. olivacea* but also for the wild untagged crabs. In a small experiment wherein 10 tagged and 10 untagged (control) crabs (50-80 mm CW) were kept for 40 days in individual compartments placed in a pond near the collection area and fed fish by-catch, no crab moulted and a 100% survival was observed in both groups. These findings indicate that the microwire tags used had no effect on the growth, moulting and survival of crabs. The results of the general linear regression model both for length and weight are shown in Figure 4.7a and 4.7b. In general, crabs in this particular experiment grow at the rate of 0.25 mm CW d^{-1} and 1 g BW d^{-1}.

Analysis of variance was used in comparing growth rates within size classes both in length and weight, length between sexes and weight between sexes. There were significant differences in size classes in length ($F=4.77$, $p<0.001$). Crabs measuring 51-60 mm CW showed a significantly higher growth rate (ANOVA, $T=2.65$, $p<0.05$) than the average while those measuring 61-70 mm CW a significantly lower growth rate (ANOVA, $T=-1.99$, $p<0.05$). Although crabs <41 mm showed the highest and those >91 mm the lowest growth rates, they were not significant due to the small number of samples (Fig. 4.8a). Table 4.2 shows the result of the regression analysis (slopes and intercepts of each length size class compared), the analysis of variance table and the comparisons between length size class regression slopes and intercepts with the respective group averages. There was no significant difference in growth rates between weight size classes ($F=0.17$, $p>0.05$) (Fig. 4.8b). This was further supported by Levene's Test on residual variances showing no significant differences between the four weight size classes (Test statistic=2.11, $p>0.05$).

Moreover, it was observed that growth between release and recapture constantly decreases with increasing length and weight size classes, respectively. All crabs

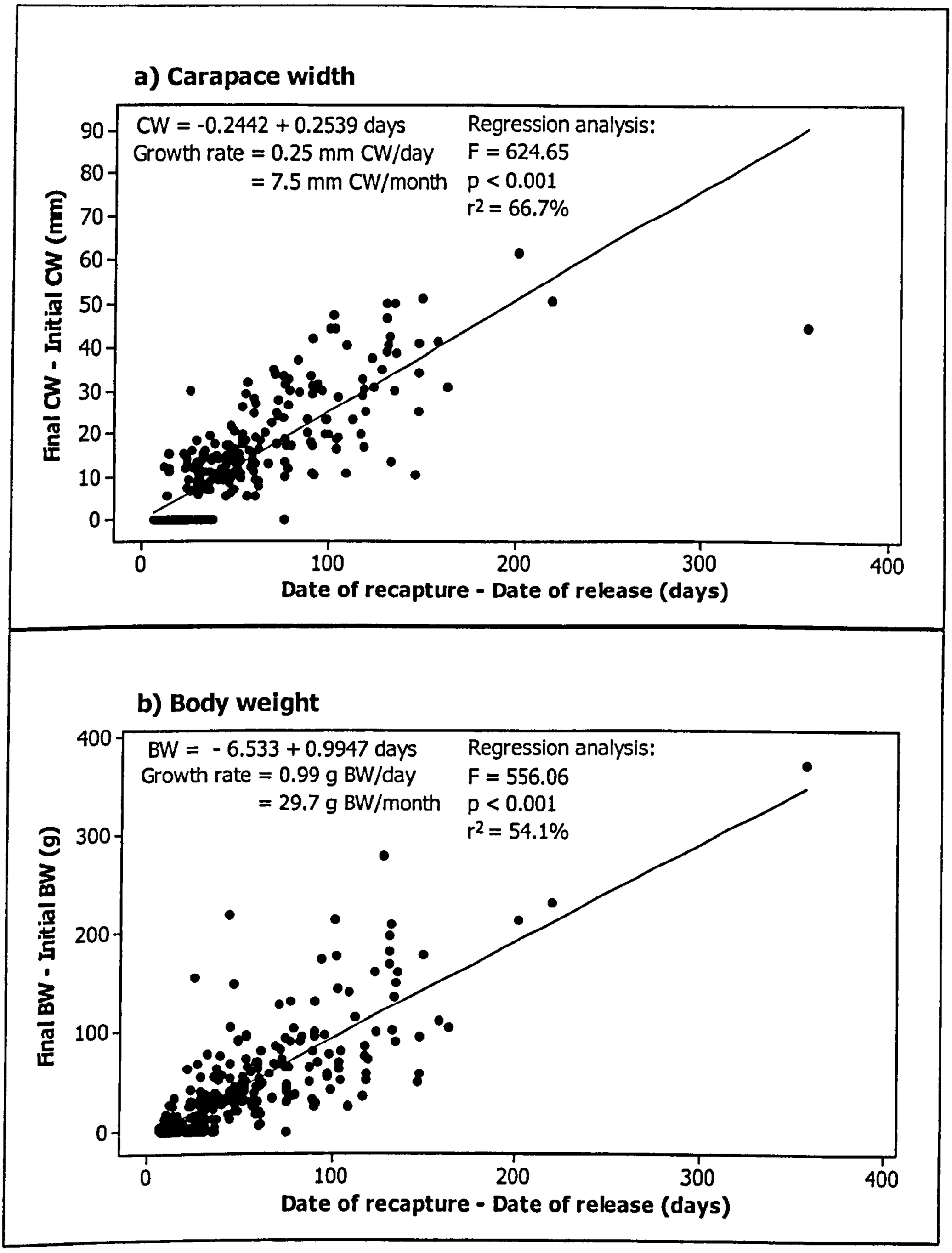


Figure 4.7. Growth rates of tagged *Scylla* spp. in terms of a) carapace width and b) body weight from date of release to date of recapture in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, February 2004-February 2005. Growth rate values were obtained from the slope of the regression line.

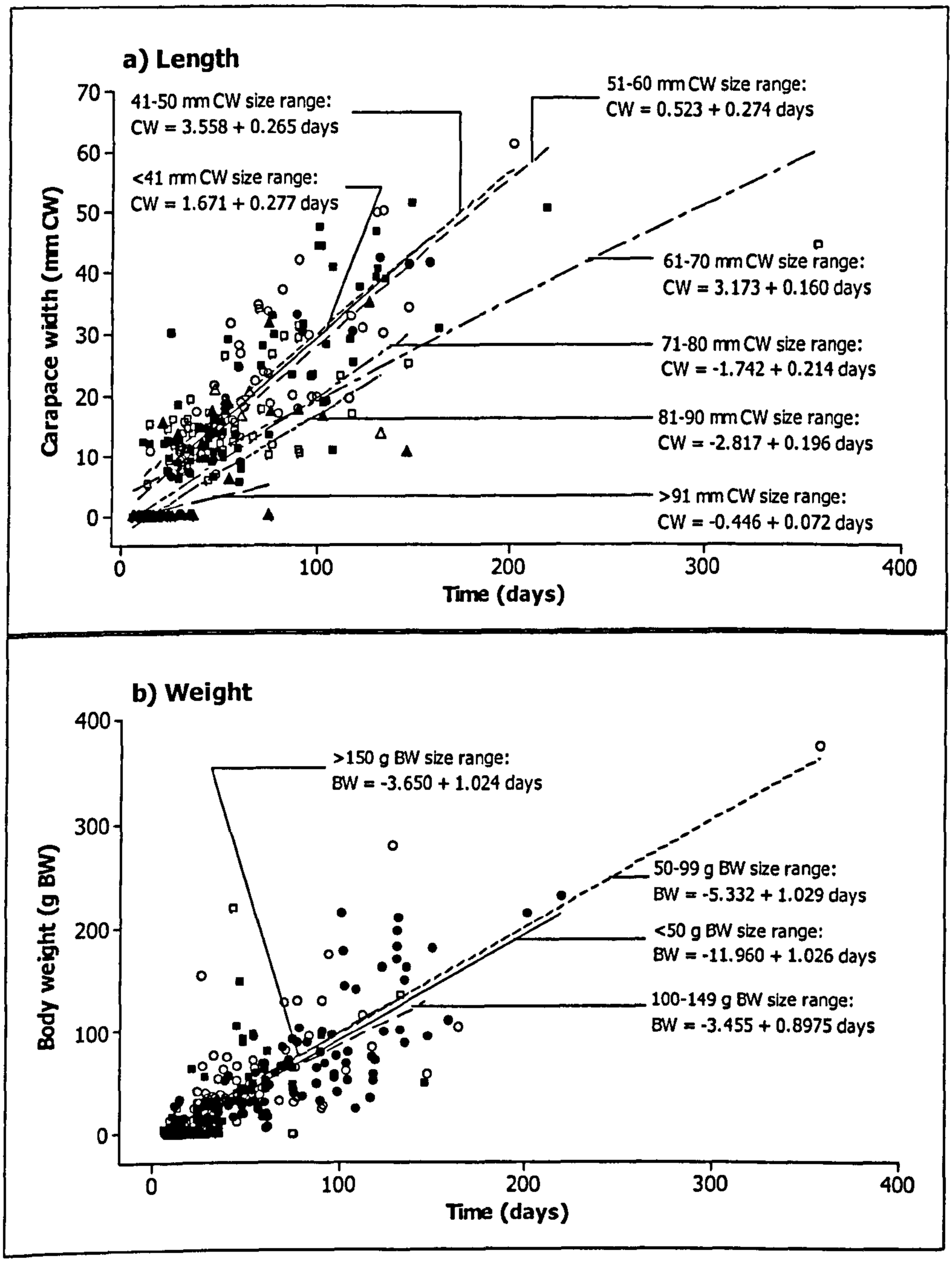


Figure 4.8. Comparison of growth rates of *Scylla* spp. from different size classes of a) length and b) weight between release and recapture in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, February 2004-February 2005. Filled circles = <41 mm and <50 g, empty circles = 41-50 mm and 50-99 g, filled squares = 51-60 mm and 100-149 g, empty squares = 61-70 mm and >150 g, filled triangles = 71-80 mm, empty triangles = 81-90 mm, * = >91 mm; a line connects the regression line to the equation for each size class.

Table 4.2. Analysis of carapace width (mm CW) increase in different size classes of *Scylla* spp. between days of release and recapture (time).

a) Regression analysis of the increase in mm CW with time

Size class	Slope ± S.E (mm d^{-1})	Intercept ± S.E (mm)
<41 mm	0.277±0.06	1.671±5.38
41-50	0.265±0.03	3.558±2.12
51-60	0.274±0.02	0.523±1.59
61-70	0.160±0.02	3.173±1.40
71-80	0.214±0.03	-1.742±1.59
81-90	0.196±0.05	-2.817±1.91
>91 mm	0.072±0.11	-0.446±2.88

b) Analysis of variance table with time in days as covariate; Seq=sequential, Adj=adjusted for entry order into the model

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Size class	6	16807.0	679.5	113.3	2.49	0.023
Days	1	22704.8	5073.3	5073.3	111.34	<0.001
Size class*Days	6	1303.9	1303.9	217.3	4.77	<0.001
Error	300	13670.1	13670.1	45.6		
Total	313	54485.7				

c) Comparisons between size class regression slopes and intercepts and their respective group averages. Slope differences in mm CW d^{-1}, intercept differences in mm. P values with * are significantly different from the average.

		Average	Difference	T-value	P
Size class	Slopes	0.208±0.04			
<41 mm			0.068	1.33	0.185
41-50			0.057	1.92	0.055
51-60			0.066	2.65	0.008*
61-70			-0.048	-1.99	0.048*
71-80			0.006	0.18	0.854
81-90			-0.012	-0.26	0.793
>91 mm			-0.137	-1.49	0.138
Size class	Intercepts	0.56±2.41			
<41 mm			1.111	0.21	0.836
41-50			2.998	1.41	0.158
51-60			-0.037	-0.02	0.981
61-70			2.613	1.86	0.063
71-80			-2.302	-1.45	0.149
81-90			-3.376	-1.76	0.079
>91 mm			-1.006	-0.35	0.727

measuring <41 mm during release showed 100% increase in length upon recapture compared with only 8.3% of the crabs >91 mm CW. There was 94.1% growth in crabs weighing <50 g and only 55.6% in crabs >150 g BW.

ANOVA further showed that growth rate in terms of length between male and female crabs significantly differ from each other ($F=15.62$, $p<0.001$). The growth rates were 6.6 mm and 9.0 mm CW month^{-1} for males and females, respectively (Fig. 4.9a). Table 4.3 shows the result of the regression analysis (slopes and intercepts of each sex compared for length), the analysis of variance table and the comparisons between sex regression slopes and intercepts with the respective group averages. Although males grew faster at 32.1 g than females at 27.3 g BW month^{-1} (Fig. 4.9b), ANOVA showed that growth rates do not significantly differ between sexes ($F=3.52$, $p>0.05$). This was further supported by Levene's Test on residual variances showing no significant differences between sexes (Test statistic=3.54, $p>0.05$).

Derivation of maximum size was done using the Gulland-Hold plot which was applicable only to length. Based on this method, L_∞=148.02 mm CW while K=0.002774 (Fig. 4.10). Using the Beverton's method, L_∞=147.37 mm CW. Since both values obtained for length are almost equal and since the Gulland-Holt plot can't be applied to weight data, W_∞ was obtained using Beverton's method (W_∞=652.63 g BW). These values apply only to *S. olivacea* since L_∞ and W_∞ were estimated from *S. olivacea* samples (n=7,620) obtained for 43 months (April 2002-November 2005) for the population and fisheries study (Chapter 3). The maximum size (L_∞ and W_∞) obtained using Beverton's method may also represent estimates for the male population since the crabs used for obtaining these estimates were male. Using Beverton's method, L_∞ and W_∞ for female *S. olivacea* population are 131.58 mm CW and 463.16 g BW, respectively.

Population estimate, mortality and survival

Release (sessions 1-6) and recapture data (sessions 2-26) until the last marked crab was recaptured are shown in Table 4.4. Of the 822 crabs released, 38.20% were recaptured. The highest percentage recovery of 46.51% was from the last release session and the lowest from session 1 at 29.47%.

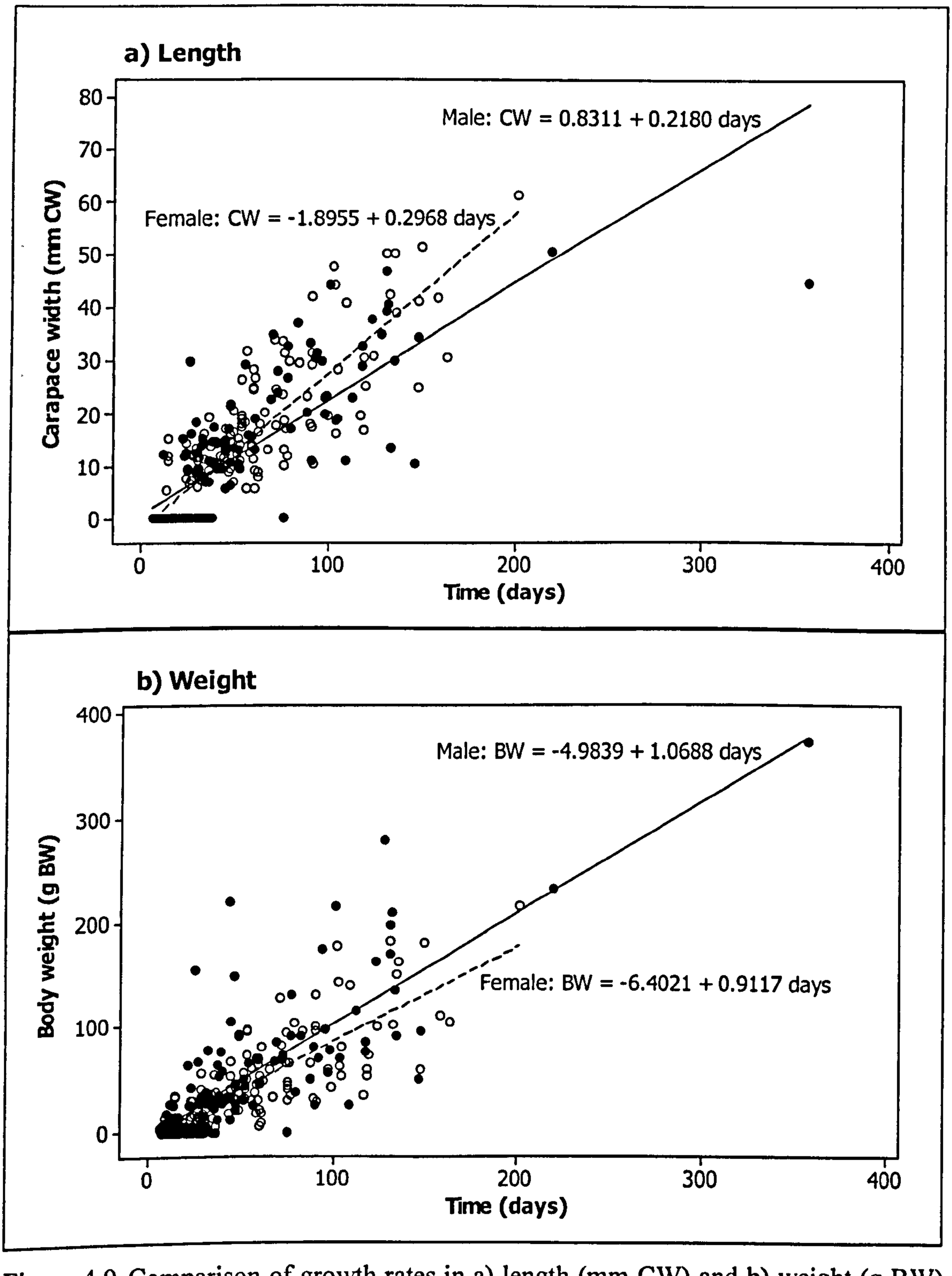


Figure 4.9. Comparison of growth rates in a) length (mm CW) and b) weight (g BW) of male and female *Scylla* spp. between release and recapture in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, February 2004-February 2005. Filled circles = male, empty circles = female; solid line = male regression line, broken line = female regression line.

Table 4.3. Analysis of carapace width (mm CW) increase in *Scylla* spp. males and females between days of release and recapture (time).

a) Regression analysis of the increase in mm CW with time

Sexual development	Slope ± S.E (mm d^{-1})	Intercept ± S.E (mm)
Female	0.297±0.01	-1.896±0.64
Male	0.218±0.01	0.831±0.64

b) Analysis of variance table with time in days as covariate; Seq=sequential, Adj=adjusted for entry order into the model

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Sex	1	249	250	250	4.51	0.035
Days	1	36180	36942	36942	665.18	<0.001
Sex*Days	1	866	866	866	15.62	<0.001
Error	310	17191	17191	55		
Total	313	54486				

c) Comparisons between sexes regression slopes and intercepts and their respective group averages. Slope differences in mm CW d^{-1}, intercept differences in mm. P values with * are significantly different from the average.

		Average	Difference	T-value	P
Sex	Slopes	0.257±0.01			
Female			0.039	3.95	<0.001*
Male			-0.039	-3.95	<0.001*
Sex	Intercepts	-0.532±0.64			
Female			-1.363	-2.12	0.035*
Male			1.363	2.12	0.035*

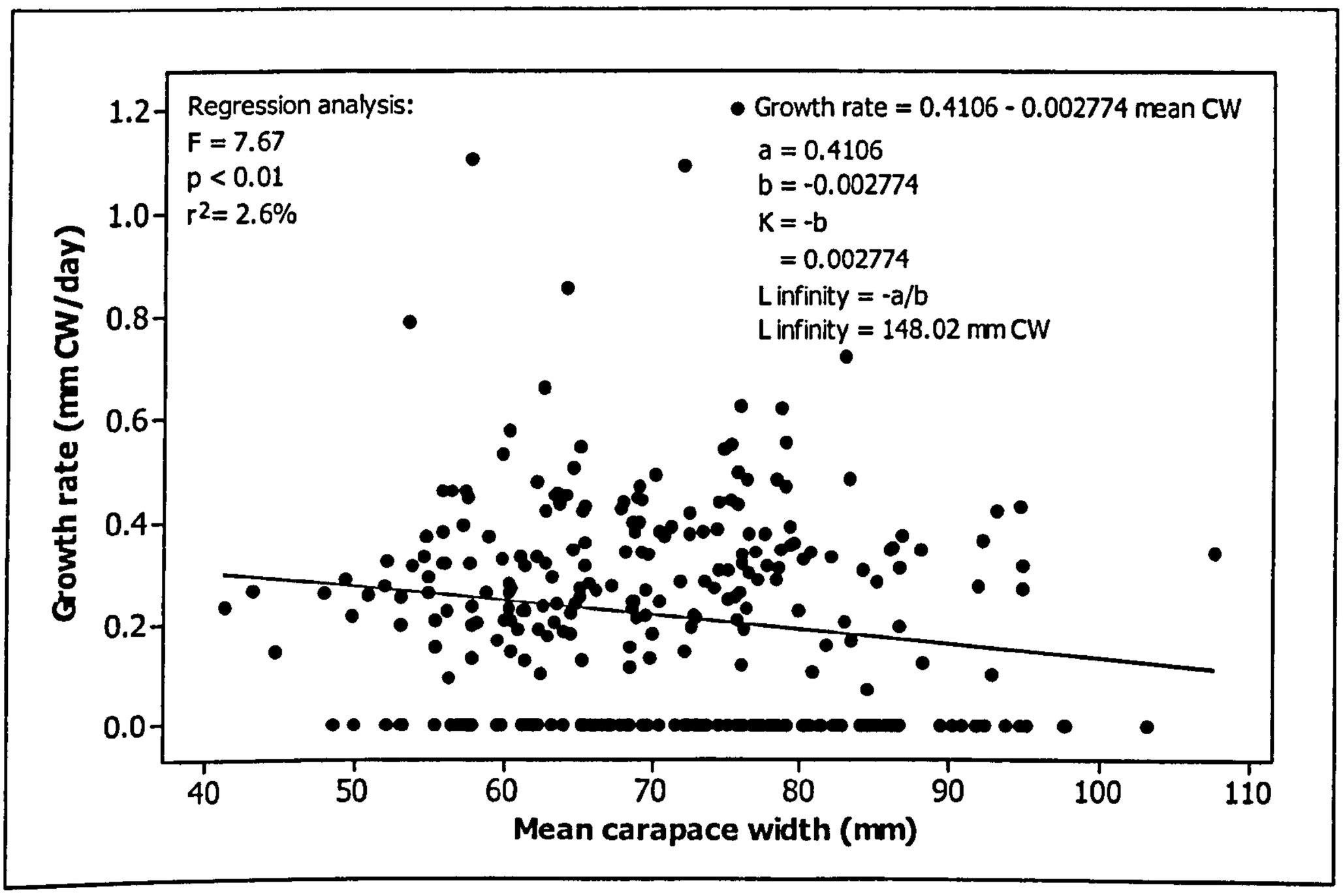


Figure 4.10. Gulland-Holt plot used to estimate the asymptotic length (L_∞) and K of *Scylla* spp. released and recaptured in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, February 2004-February 2005.

Table 4.4. Tag returns from marked *Scylla* spp. from each of the release sessions conducted in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines.

Session Date	Number Released	Recoveries from each session representing one spring tide																										Total	Percent recovery
		2	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**			
1 (5-8 Feb)	95	13	6	3	0	0	2	1	0	2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0		28	29.47
2 (19-22 Feb)	171		21	13	10	4	4	5	3	3	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	65	38.01
3 (5-8 Mar)	126			23	6	5	3	5	2	1	2	2	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	52	41.27
4 (18-21 Mar)	143				13	14	10	3	5	4	1	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	52	36.36
5 (11-14 Apr)	158					17	10	8	7	6	2	3	2	1	0	0	0	0	1	0	0	0	0	0	0	0	0	57	36.08
6 (20-23 Apr)	129						18	12	7	8	3	0	3	3	3	1	1	0	0	0	1	0	0	0	0	0	0	60	46.51
Total	822	13	27	39	29	40	47	34	24	24	8	7	10	4	3	1	1	0	1	0	1	0	0	0	0	1		314	38.20

Results of the Jolly-Seber analysis using Simply Tagging software are summarized in Table 4.5. Population estimates ranged from 691 to 1,684 crabs. An increasing trend was observed from session 2 to session 5 then decreasing in the following session. Population estimate in session 5 (N=1,684) was significantly higher than all the other four sessions (Fig. 4.11). Using a 50-ha mangrove area (Chapter 2), this gives a crab density of 14 crabs ha^{-1} in session 2 to 34 crabs ha^{-1} in session 5, or 1 crab for every 715 and 295 m^2, respectively. The relative abundance of the population, as shown by the CPUE, was parallel to the population estimates from this mark-recapture study. Although there were no significant differences in CPUE between sessions, both CPUE and population estimates followed the same trend except in session 2 when CPUE was falling and population estimate was rising (Fig. 4.12). Mean CW of crabs significantly decreased (ANOVA, $F=5.72$, $p<0.001$) from session 2 to session 6 which may reveal an increase in number of smaller crabs (Fig. 4.13) during the period of the study. Jolly-Seber analysis also revealed an addition of 1,545 new animals from session 2 to 6 (Table 4.5, Fig. 4.13). Hence, the increase in population may be attributed to recruitment. The highest abundance of crabs in session 5 may be due to the high survival estimate of new animals or recruits in session 4, which was the session with the highest recruitment among the 6 sessions (Table 4.5). Between sessions 1 and 2 the population decreased by 45% due to mortality and emigration ($1 - \Phi$) which further decreased to 23% between session 2 and 3, 17% between sessions 3 and 4, and 14% between sessions 4 and 5. These decreasing mortality and emigration rates, coupled by the addition of new recruits every session, have caused the increasing population estimates from sessions 2 to 5. The drop of population in session 6 may be attributed to the increase in mortality and emigration rates to 35% and the decrease in the number of recruits in session 5.

Movement

In general, the results of this study showed that most of the crabs tend to stay close to where they were released or to have very minimal movement. The highest percentage of recapture was usually in areas where they were released and that they stayed in these areas up to a maximum of 163 days, as in the case of the crabs released and recaptured in Site 2 (Table 4.6). Figure 4.14 shows the percentages of recapture in six

Table 4.5. Population, recruitment and survival estimates of *Scylla* spp. in six release sessions (dates) in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines using the Jolly-Seber model for open populations and analysed in Simply Tagging software.

Release Session (dates)	Proportion of recaptures	Marked animals in population	Population estimate	S.E. of pop estimate	Survival estimate	S.E. of survival	New animals	Probability of capture
1 (5-8 Feb)		0			0.550	0.090		
2 (19-22 Feb)	0.076	52.3	690.9	189.1	0.771	0.102	398.4	0.268
3 (5-8 Mar)	0.175	161.3	920.2	185.7	0.832	0.117	251.6	0.167
4 (18-21 Mar)	0.220	217.4	994.7	171.7	0.862	0.122	860.9	0.184
5 (11-14 Apr)	0.164	276.2	1,684.0	328.6	0.675	0.084	34.2	0.112
6 (20-23 Mar)	0.238	272.2	1,150.0	186.8	Not Def.	Not Def.	0.0	0.147

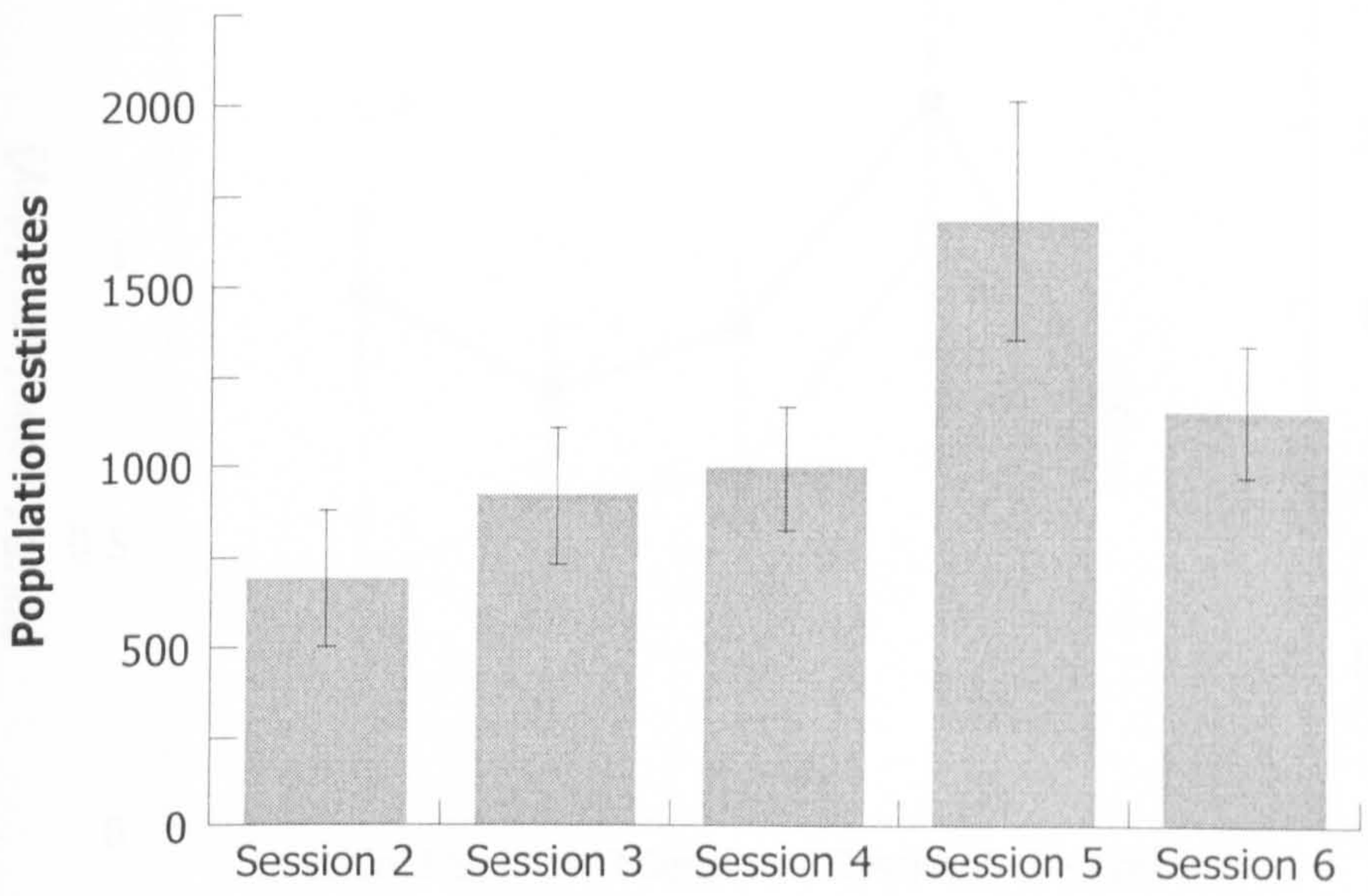


Figure 4.11. Population estimates from mark-recapture data of *Scylla* spp. tagged, released (February-April 2004) and recaptured (February 2004-February 2005) in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines.

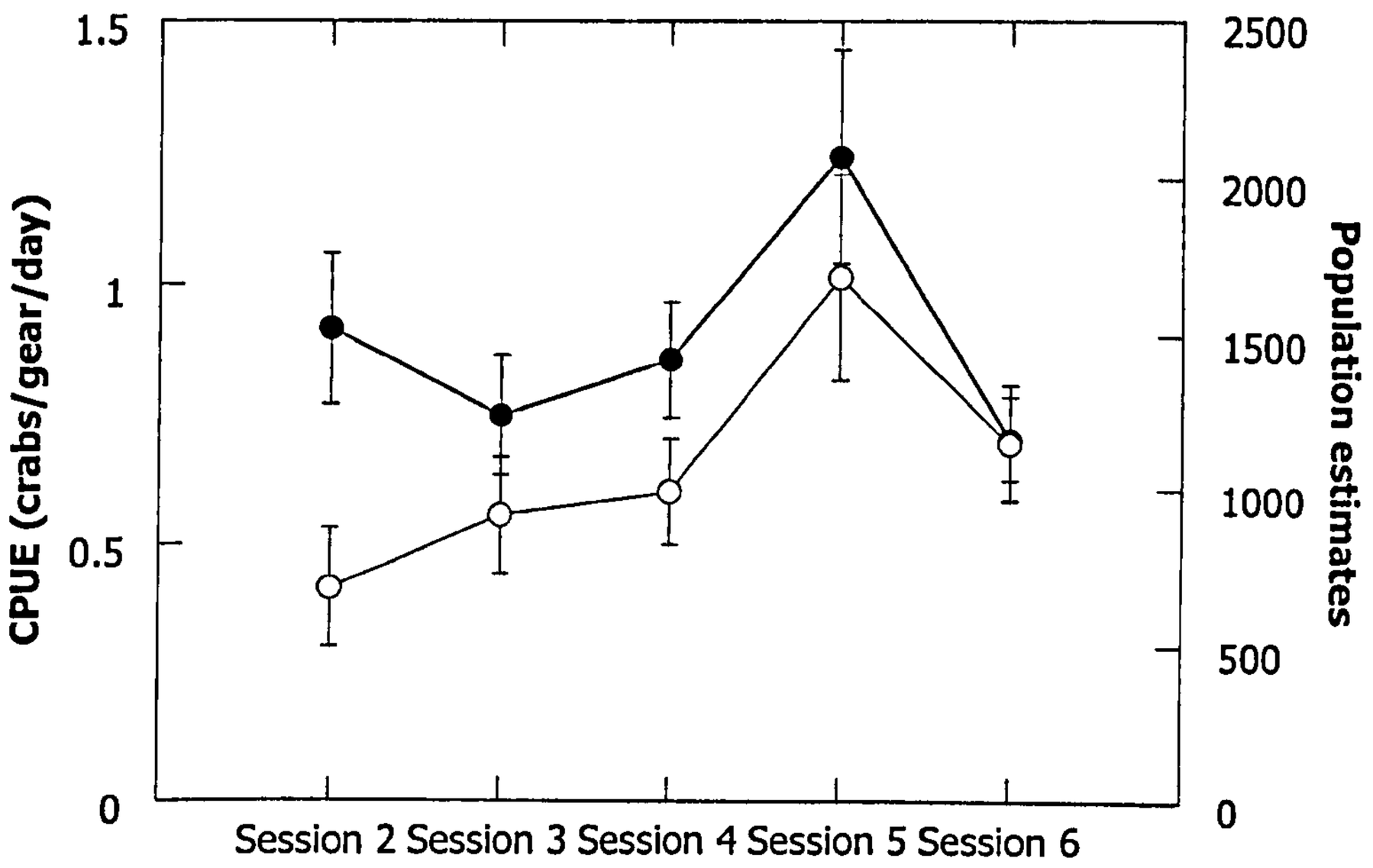


Figure 4.12. Catch per unit effort (filled circles) during the sessions when population estimates (empty circles) were obtained from the mark-recapture data of *Scylla* spp. tagged, released (February-April 2004) and recaptured (February 2004-February 2005) in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines.

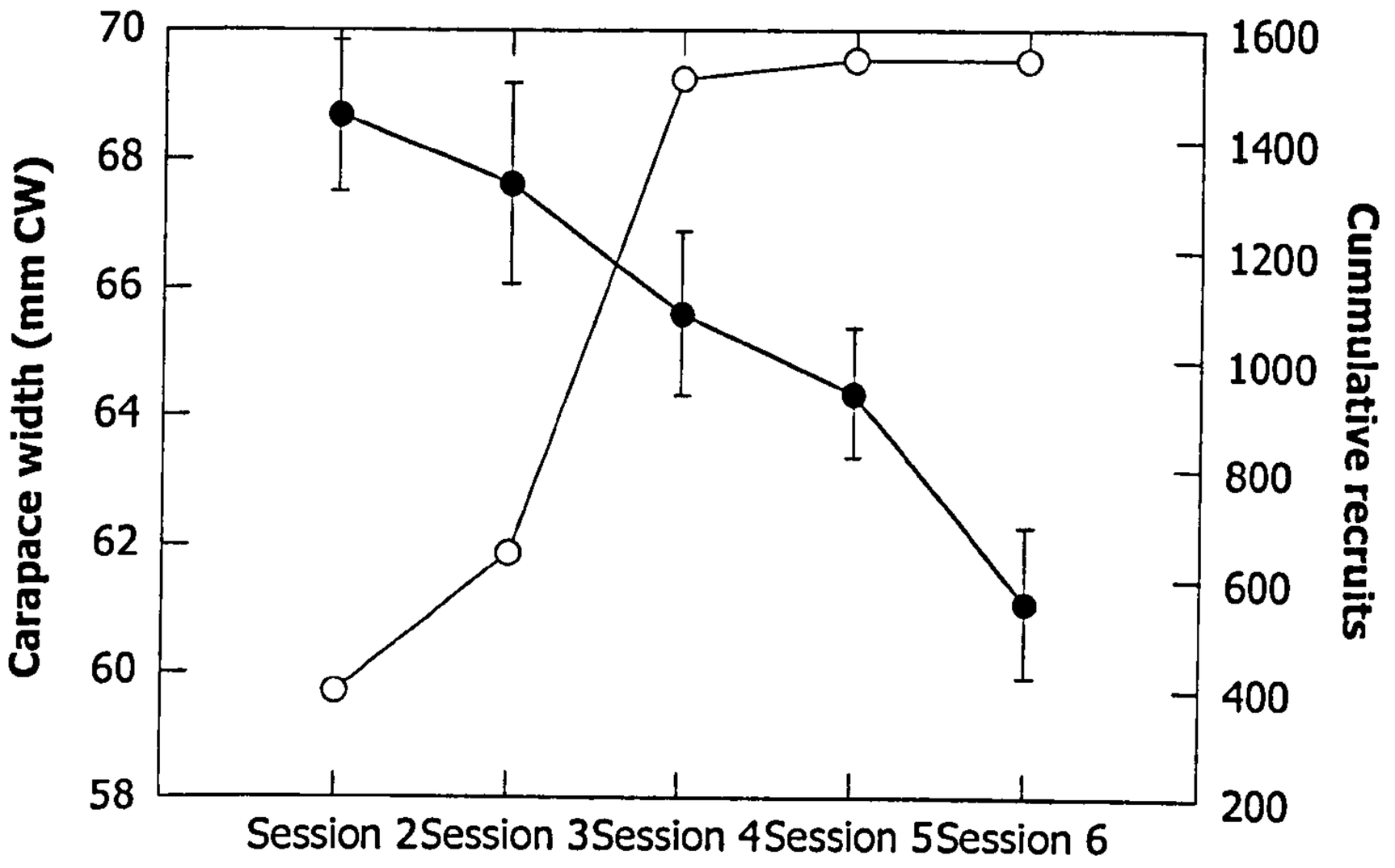


Figure 4.13. Mean carapace width (mm CW) (filled circles) and cumulative recruits (empty circles) during the sessions when population estimates were obtained from the mark-recapture data of *Scylla* spp. tagged, released (February-April 2004) and recaptured (February 2004-February 2005) in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines.

Table 4.6. Period of movement of crabs from release site to recapture areas in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines. Numbers in parenthesis below release sites denote total number of crabs released from that site in all 6 sessions; N denotes number of crabs captured from a given collection area.

Release site	Collection area	Duration (days)	N
Site 1	Site 1	9 – 92	44
(306)	Site 2	no recaptures	0
	Site 3	no recaptures	0
	Site 4	no recaptures	0
	Site 5	9 – 147	32
	Site 6	10 – 55	9
Total			85
Site 2	Site 1	357	1
(198)	Site 2	67 – 163	14
	Site 3	13 – 219	16
	Site 4	8 – 55	4
	Site 5	7 – 158	33
	Site 6	no recaptures	0
Total			68
Site 3	Site 1	no recaptures	0
(154)	Site 2	11 – 83	4
	Site 3	7 – 118	43
	Site 4	9 – 147	16
	Site 5	6 – 89	31
	Site 6	26	1
Total			95
Site 4	Site 1	no recaptures	0
(164)	Site 2	65	1
	Site 3	6 – 132	19
	Site 4	10 – 147	42
	Site 5	10 – 41	4
	Site 6	no recaptures	0
Total			66

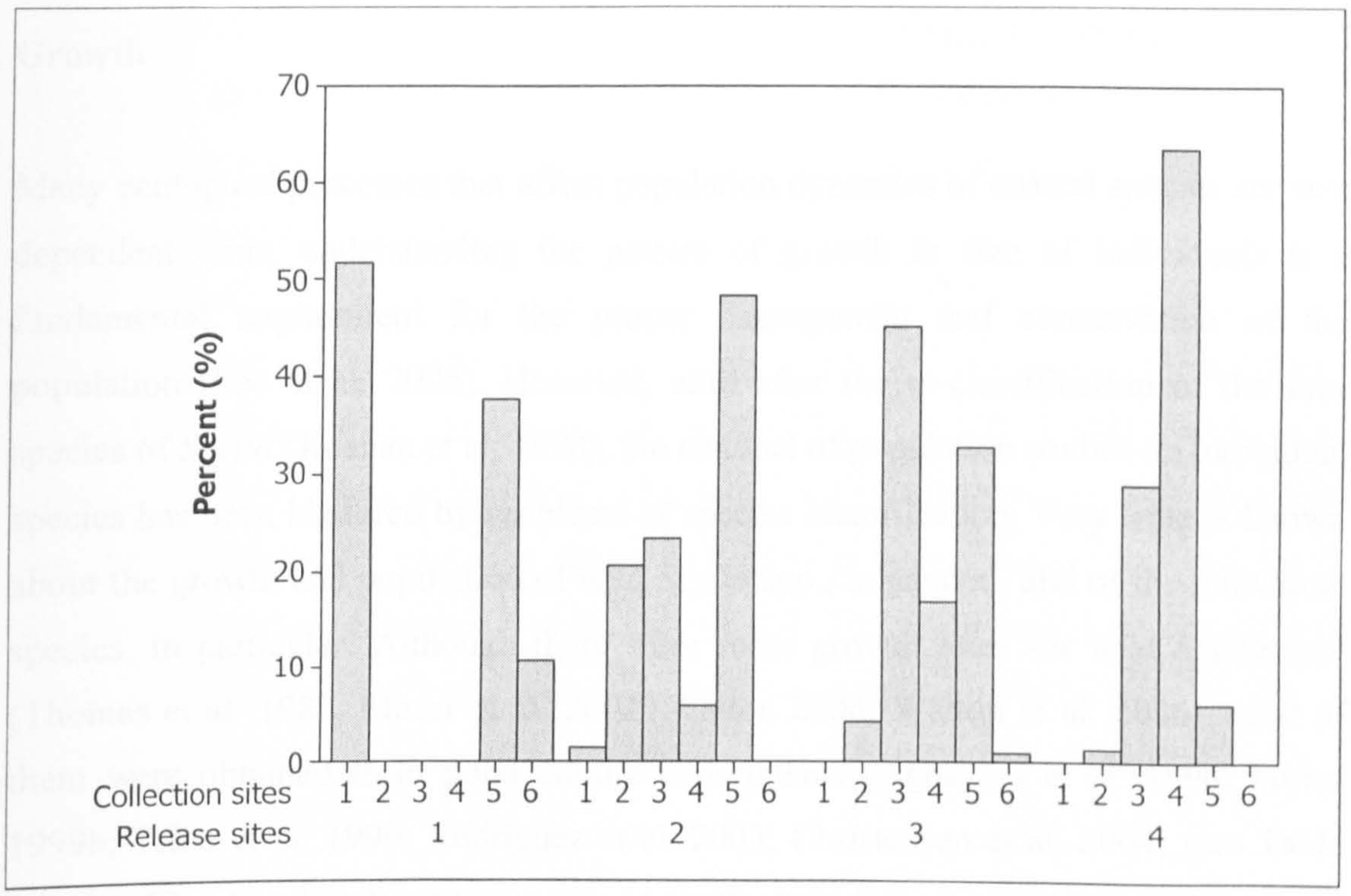


Figure 4.14. Percentage of total *Scylla* spp. recaptured from different collection areas in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; February 2004-February 2005. Release sites are shown below each cluster of collection areas. Refer to Table 4.6 for total number of recaptures from each collection site (N).

different collection areas of crabs released in four different release sites. As high as 60% of the recaptures (Site 4) were in the vicinity of the release site. In Site 2, however, almost 50% of the recaptures were caught in Site 5, an area very close and directly connected by a major river branch to Site 2 (see Fig. 4.1).

DISCUSSION

Growth

Many ecological processes that affect population dynamics of coastal species are size dependent, thus, understanding the pattern of growth in size of individuals is a fundamental requirement for the proper management and conservation of the population (Lee et al. 2006). However, until after the re-classification of the four species of *Scylla* (Keenan et al. 1998), the conduct of population studies on individual species has been hindered by problems of species identification. Very little is known about the growth and population of wild *Scylla* spp., in general, and of the individual species, in particular. Although there were some growth rates for wild *Scylla* spp. (Thomas et al. 1987; Moser et al. 2002; Walton 2006; Walton et al. 2006), most of them were obtained from pond culture experiments (Agbayani et al. 1990; Fortes 1999b; Trino et al. 1999; Rodriguez et al. 2003; Christensen et al. 2004) (see Table 1.3 in Chapter 1). In crustaceans, growth has been quantified with several measurements of body size and weight and relationships between these characteristic data have been described by linear or nonlinear models (Anger and Moreira 1998). For growth to occur, crustaceans like the mud crabs must undergo periodic ecdysis, the shedding of the confining exoskeleton (Zou and Fingerman 1999). From the recovered marked crabs, a growth rate in CW of 0.25 mm d^{-1} was obtained using the general linear regression model. Since growth in crustaceans in terms of length is discontinuous and mainly depends on moulting, this rate does not imply that the crab grows this much length per day. Increase in length may vary depending on the duration of the moult cycle, or on the size and sex of crabs as shown in the results of this present work. In terms of weight, crabs grow at the rate of 1 g d^{-1}. Body weight

does not depend on moulting, hence growth in terms of weight may be considered continuous.

Most pond studies have generated a variety of growth rates in different species of *Scylla*. However, studies before Keenan et al.'s (1998) report when species identification was ambiguous will be reported in this present work as *Scylla* spp. Since aquaculture is more concerned with weight which is equated to production, than with length, growth rates in these studies are mostly in terms of weight gain per day. In a mixed-culture of *S. olivacea* and *S. paramamosain*, a daily weight gain of 2 g was reported by Christensen et al. (2004). A higher growth rate was observed by Agbayani et al. (1990) for *Scylla* spp. at 1.62-2.28 g d^{-1}. On the other hand, Fortes (1999b) reported growth rates of 0.69-1.28 g BW d^{-1} and 0.12-0.25 mm CW d^{-1} in *S. olivacea* grown in brackish water earthen ponds at different densities and in the presence or absence of shelters. In another pond experiment where *S. olivacea* were fed twice daily with different diets, growth rates ranged 1.22-1.37 g BW d^{-1} (Rodriguez et al. 2003). The growth rates obtained in the present study, dominated by *S. olivacea* and may be considered to represent the species, are similar to those obtained for *S. olivacea* in the pond experiment of Fortes (1999b). The differences in growth rates in ponds due to density and presence or absence of shelter (Fortes 1999b) or due to differences in diets (Rodriguez et al. 2003) may also be some of the factors that affect growth rates of crabs in the wild.

It was observed that growth between release and recapture decreases with increasing size. There was 100% growth in recaptured crabs measuring <41 mm and only 8.3% in crabs >91 mm CW, while 94.1% in crabs weighing <50 g and only 55.6% in those crabs >150 g. However, significance in growth rates between size classes was observed only in length but not in weight, showing that weight is continuous and does not depend on moulting. In the spider crab *Pyromaia tuberculata*, Flores et al. (2002) reported a decreasing percentage in moult increment with sexual maturity. Crabs measuring 51-60 mm CW showed a significantly higher growth rate than the average (0.21 mm d^{-1}) while those measuring 61-70 mm CW a significantly lower growth rate. Although crabs <41 mm showed the highest and crabs >91 mm the lowest growth rates, they were not significant due to small number of samples (Fig. 4.8a). Even though considered the fastest growing species of *Scylla*, growth rate of *S. serrata* does

not differ much with *S. olivacea* in smaller size crabs. In laboratory conditions, growth rate of *S. serrata* from a mean size of 52.3 to 65.7 mm CW ranged 0.25-0.39 mm d^{-1} while those from 65.7 to 79.3 mm CW ranged 0.22-0.27 mm d^{-1} (Catacutan 2002). Crabs undergo several moults during their lifetime but moult frequency and moult increment were greater for the early crab stages and decreased in the later stages (Arriola 1940; Ong 1966). Thus, the decreasing growth rates as the crabs grow bigger or mature, as also observed in this study. In *Callinectes sapidus*, for example, the average moult cycle was 37 days with the moult interval increasing as the animals grow; ranging from 30 days in the smaller juveniles to 47 days for the larger mature ones (Freeman et al. 1987). In *S. serrata* grown in the laboratory, Catacutan (2002) reported that the duration of moulting in crabs varies between 34-54 days in the smaller sizes and between 50-61 days for the larger ones.

The result of the study further showed that females significantly grew faster in length (0.30 mm CW d^{-1}) than males (0.22 mm CW d^{-1}). The same was observed by Ut (2002) in *S. paramamosain* wherein crabs recaptured between 10-44 days after release had higher growth rates in females (0.94 mm d^{-1}) than in males (0.82 mm d^{-1}). However, at an extended period of 144 days in the wild, males (0.70 mm d^{-1}) tended to grow faster than females (0.68 mm d^{-1}). Although no significant difference was observed between sexes, males showed a higher growth rate in weight (1.07 g BW d^{-1}) than females (0.91 g BW d^{-1}). Manganpa et al. (1987) in Cholik (1999) concluded that male crabs grew faster than females. *Scylla* spp. males grew at an average growth rate of 1.30 g d^{-1}, while the females grew only 0.90 g d^{-1}. Trino et al. (1999) reported much higher growth rate in a mono-sex culture trial of mixed *S. serrata* and *S. tranquebarica*, with males growing at the rate of 3.90 g d^{-1} and the females at 2.60 g d^{-1}.

L_∞ was derived from two different methods and the values obtained were almost similar. Using the Gulland-Hold plot L_∞=148.02 mm CW and K=0.002774 while using Beverton's method, L_∞=147.37 mm CW. W_∞ obtained using Beverton's method was W_∞=652.63 g BW. Christensen et al. (2004) obtained an almost similar value of L_∞=145 mm CW from the mixed culture of *S. olivacea* and *S. tranquebarica* using the von Bertalanffy growth equation assuming an age of 2 months at the start of the

culture period. In the present study, however, age could not be assumed since samples were obtained from the wild, thus age and sizes of released crabs vary.

Moreover, it was observed that microwire tags used had no effect on the growth, moulting and survival of crabs. Comparing tagged and untagged *S. paramamosain*, Ut (2002) reported no significant difference in the intermoult duration in crabs kept in individual containers in concrete tanks, no significant difference in the specific growth rates in crabs kept in individual cages in ponds, and no significant difference in final weight in free-ranging crabs in ponds. Similar results were reported by Fitz and Wiegert (1991), Davis et al. (2004) and Eggleston et al. (2004) for *Callinectes sapidus*.

Population estimate, mortality and survival

The use of mark-recapture method to get an estimate of the population had been utilized in many studies involving marine animals both vertebrates and invertebrates. Bennett (1979) and Bell et al. (2003) have worked on *Cancer pagurus*; Diaz and Conde (1989) for *Aratus pisonii*; van Montfrans et al. (1991), Fitz and Wiegert (1992) and Eggleston et al. (2004) and references therein for *Callinectes sapidus*. Several studies have been done on the different species of *Scylla* (Hill 1975; Williams and Hill 1982; Robertson and Piper 1991; Le Vay et al. 1999; Moser et al. 2002; Ut 2002; Walton 2006).

Population estimates can be derived using a variety of models. The Jolly-Seber model was used in the present work since it was found to be the most appropriate method given that the crab population is considered open and that crabs were recaptured only once. It provides a modified model that allows analysis of samples in such a single recapture situation (Schwarz and Seber 1999). It is the model used by Eggleston et al. (2004) for *Callinectes sapidus* because it does not assume population closure and it not just give survival values but population estimates, recruitment and capture probability, as well. The same method was used by Fitz and Wiegert (1992) in a mark-recapture study of *Callinectes sapidus* involving recapture with removal. Meeting the assumptions of capture-recapture models is critical to ensuring unbiased analysis, hence, the experiment was designed to minimize violation of the

assumptions. The tagging system is critical in this experiment and the choice of internal microwire tag assumes tag retention even after the crab has undergone several moulting. In a laboratory experiment, microwire tags were retained effectively (96-98% minimum) through multiple moults by the smallest *Callinectes sapidus* and tag presence had no effect on growth or survival rates over 80 days (Fitz and Wiegert 1991). In a study conducted by Davis et al. (2004), the same crab species showed higher tag retention with the coded microwire tags compared with the elastomer tags. Ut (2002) also reported high tag retention of 87-100% in *S. paramamosain* even after several moults. The release of crabs at an even distance along the four creeks in the study site and delaying collection of crabs by one day in nearby collection sites ensures even dispersal of the crabs in the mangroves and allows uniform mixing with the untagged population to give them the same probability of capture as those with the untagged ones. To avoid concentration in a limited area and to have an approximately even sampling of the population both of tagged and untagged crabs, collectors were assigned designated areas for collection (see Fig. 4.1). The use of bamboo trap, which based on the fisheries study is the least selective of the three gears in terms of species, sex and size (Chapter 3), gives the same probability of capture of the surviving population during sampling of both the tagged and untagged individuals. The buying of all crab catches every spring tide, irrespective of size or the presence of tagged crabs, eliminates post-capture size selection and minimizes loss of recovered tags, respectively.

Population estimates ranged from 691 (session 2) to 1,684 crabs (session 5). Using a 50-ha mangrove area, this gives a crab density of 14-34 crabs ha^{-1}. This estimate parallels the estimate of *S. olivacea* density of 21-28 crabs ha^{-1} (minimum: 14 ha^{-1}; maximum: 38 ha^{-1}) in Ranong, Thailand, a habitat which is characterized by Moser et al. (2005) as heavily fished. Despite the differences in habitat, more particularly in temperature, the result of the present work is also very much similar to the findings of Hill (1979b) for *S. serrata* in the St. Lucia system, in a south African estuary where the density was estimated at 15-33 crabs ha^{-1}. In another study of Hill (1975) in two South African estuaries, the population of *S. serrata* in a closed estuary was estimated at 1 crab 124 m^{-2} area or 80 crabs ha^{-1}. Robertson and Piper (1991) reported crab densities of *S. serrata* in two closed estuaries in Natal, South Africa at 44-53 crabs ha^{-1}. Using burrow occupancy of crabs, Nandi and Dev Roy (1990) estimated *Scylla*

spp. densities at 3,200-12,800 crabs ha^{-1} in the Hooghly estuary and 2,400-12,000 crabs ha^{-1} in the Matla estuary, West Bengal. In Utende, Tanzania, density estimates were 1,228 crabs ha^{-1} in the mangrove fringe while 324 crabs ha^{-1} in the inner mangrove forest (Barnes et al. 2002). Using cast nets and drag nets, density of mud crabs in Pichavaram mangroves in India varied with salinity, such that no crabs were caught during monsoon season between October and November when salinity was almost 0 p.s.u., 1,418 crabs ha^{-1} at post monsoon and 434 crabs ha^{-1} during other times of the year. Crab density started to increase once the salinity increased and attained maximum density at salinities between 10-20 p.s.u. (Chandrasekaran and Natarajan 1994). The density of *S. olivacea* obtained by Walton (2006) in a replanted mangrove area in Kalibo, Aklan was 36-138 crabs ha^{-1}, up to 4 times greater than the maximum density obtained in the present study. Aside from over-fishing, the low density of crabs in the study site may be attributed to the topography of the area. The mangroves in Ibajay, Aklan as described in detail in Chapter 2, are situated some 1.5 km away from the mouth of Naisud River, the only passage that connects the area to the sea. This narrow access, though an advantage at some point by limiting the entry of large predators, may be a major factor that limits recruitment, hence, the low density of crabs.

The relative abundance of the population, as shown by the CPUE, was parallel to the population estimates from this mark-recapture study. The similarity of population estimates using the Jolly-Seber method and CPUE suggests that CPUE is a good index of abundance and agrees with Le Vay et al. (2001). These findings confirm the reliability of the estimates of abundance of crabs using CPUE in the previous months obtained during the conduct of the baseline fisheries assessment (Chapter 3). The increase of population may be due to recruitment and high survival of the new animals in the population. Recruitment is revealed by the significantly decreasing mean CW of crabs from session 2 to session 6 (from actual data, Fig. 4.13) and the increasing number of new animals/recruits from session 2 to session 4 (from Jolly-Seber estimates, Fig. 4.13 and Table 4.5).

Mortality and emigration rates decreased from 45% between sessions 1 and 2 to 14% between sessions 4 and 5. This decrease in mortality and emigration rates may explain the increasing population estimates from session 2 to 5. This may further imply that

there is high survival of the new animals in the population coming from recruitment. Mortality in the present study is low compared with the mortality obtained by Ut (2002) for *S. paramamosain* which was 1.11 month^{-1}. The high mortality, according to Ut (2002), may be partly due to emigration which in the case of the Ibajay mangroves may be limited due to the isolated nature of the habitat. Moreover, according to Fitz and Wiegert (1992), mortality decreases with increasing size. Smaller crabs moult more frequently than larger ones making them more vulnerable to predation. However, as observed in the present study, the decreasing mean CW which almost paralleled the decreasing mortality contradicts this claim. Predation may be low, in this case, due to abundance of food as suggested by the minimal movement of the crabs (see next section) as explained by Hill (1979a). There are several factors to consider that may reduce survival of an individual in a marine environment, such as adverse conditions, lack of food, competition, and most importantly predation. In the present work, the high survival rate of *Scylla* spp. may be attributed to its behaviour and biology. In a closed mangrove ecosystem, like the one in Naisud and Bugtong Bato, Ibajay, Aklan where access to the sea is only through a single opening, the mud crabs are the only voracious predators of smaller crabs and shellfish in the mangroves. *Scylla* spp. also possesses a very aggressive character and cannibalistic behaviour. Unlike in the Ranong mangrove system where predators like crab-eating macaques (*Macaca fascicularis* and *M. irus*) exist (Moser et al. 2005), or in the Eastern Cape estuaries and Florida where the crabs are preyed upon by sharks (Robertson 1996; Franks and Gruber 2006), the predators of crabs in the mangroves in Ibajay, aside from humans, are larger *Scylla* spp. themselves which can be considered the top predator in the area. All human fishing effort was only by collectors working in the mark-recapture study.

Movement

The distance travelled by crabs could not be precisely determined from this experiment, since they were not released on one point of the site but were distributed evenly along the creek to evenly mix with the wild population in order to get a better estimate of the population density. However, a general pattern of movement can at least be obtained from the results. Generally, it was observed that most of the crabs tend to stay in the area where they were released or exhibit very minimal movements

within the mangroves. The highest percentage of recapture was usually in areas where they were released or in the adjacent collection area and that they stayed in these areas up to a maximum of 163 days, as in the case of the crabs released and recaptured in Site 2 (Table 4.6). Although almost 50% of the crabs released in Site 2 were recaptured in Site 5, these two areas are very close and directly connected with each other by a major river branch. In a mark-release-recapture study of *S. serrata* in southern African estuaries, Hill (1975) found out that *S. serrata* does not undertake much movement. In another study employing the use of transmitters to track crab movements, Hill (1978) reported that although *S. serrata* did not occupy a distinct territory, they tend to remain in the same general area despite the capability of moving at least 800 m along the length of the estuary at night. In another tagging experiment conducted on *S. serrata* using three different types of habitat in Moreton Bay, Queensland, Hyland et al. (1984) reported two categories of crab movement, such that crabs in a narrow creek with mangrove-covered banks displayed little movement while those in areas with large intertidal flats bare of mangroves underwent more movement. Also, recaptures were observed to exchange between populations in the mangrove creek and the neighbouring bay, limited exchange between the estuary and the adjacent bay and no exchange between neighbouring areas separated by a habitat unsuitable for *S. serrata*. Moreover, Ut (2002) in a mark-recapture study of *S. paramamosain*, revealed that majority of crabs remained in the study area, with the exception of crabs caught 5.8 and 12.3 km away from the release area. Considering the report of Hill (1979a) to be applicable in the present experimental setting, that crabs occur more frequently in areas where they have the highest number of prey organisms, then it can be assumed that there is enough food for the crab population in Ibajay mangroves as shown by their limited movement, regardless of the differences in the topography of the release sites (Chapter 2). Food abundance may further explain the estimate of high survival rate since cannibalism of the smaller *Scylla* spp. is minimized.

CONCLUSION

This chapter describes the application of the use of tags in a mark-recapture study for estimating population, growth and movement of the crabs within a mangrove area and the use of Jolly-Seber model for analysing open population for estimates of parameters such as population, survival/mortality and recruitment. The use of mark-recapture method as a tool in estimating the population has been utilized on *Scylla* spp. (Hill 1975; Williams and Hill 1982; Robertson and Piper 1991; Le Vay et al. 1999; Moser et al. 2002; Ut 2002; Walton 2006). In the Philippines, where a much needed assessment of this very valuable, economically important species is necessary, no work of this nature has been done before. Results of this population assessment study will be useful for future resource management programmes for the highly depleted wild stock of *Scylla* spp. This work was conducted simultaneously with Walton (2006) allowing some comparisons between *Scylla* populations in the natural mangroves in Ibajay and the replanted mangroves in Kalibo. From the population estimate data, crab density ranging from 14 to 34 crab ha^{-1} was obtained for the Ibajay mangroves. The density of *S. olivacea* (36-138 crabs ha^{-1}) obtained by Walton (2006) in a replanted mangrove area in Kalibo, Aklan was almost 4 times greater than the maximum density obtained in the present study. This shows a very low density of crabs at the present study site, comparable to the density obtained by Moser et al. (2005) for a heavily fished area in Ranong, Thailand.

Growth rates of recaptured tagged crabs were 0.25 mm CW d^{-1} and 1 g BW d^{-1}. These results were comparable to growth rates of untagged *S. olivacea* both in the wild (0.25 mm d^{-1}) (Thomas et al. 1987) and in pond culture (0.12-0.25 mm d^{-1} and 0.69-1.28 g d^{-1}) (Fortes 1999b). This suggests that the coded microwire tags do not affect growth of crabs, particularly of *Scylla* spp. and agrees with the finding of Ut (2002). Therefore, it can be used in future experiments without fear of affecting growth rates or survival of crabs.

Knowledge of variations in the features of the population is important in the conservation of exploited populations. According to Lipcius and Stockhausen (2002), some of the key demographic variables and parameters include population abundance,

recruitment relationships, size structure of the population and mean size at maturity of females (L_{mat}) which determines spawning stock. These parameters have been determined in Chapter 3 and the present chapter and may be useful in the resource management of this population. The results of the present study showing that the crab population is low, coupled with limitations in recruitment due to the limited access of the mangrove area to the sea makes stock enhancement a good approach to increasing mud crab yield. The presence of at least 50 ha of mangroves makes the area suitable for stock enhancement since totally degraded areas are unsuitable for such activity (Blankenship and Leber 1995). The acceptability of CWT to crabs thereby not affecting growth allows the use of these tags for easy monitoring of crab releases. The limited movement of crabs may suggest abundance of food resources (Hill 1979a) and less predation in the area thus minimizing emigration. All these factors combined leads to a conclusion that the mangroves in Naisud and Bugtong Bato, Ibajay, Aklan are suitable for stock enhancement trials of *Scylla* spp. When implemented, this may be an initial step leading to a more sustainable *Scylla* fishery in Ibajay, Aklan and eventually, in the Philippines.

CHAPTER 5

Stock enhancement trials

INTRODUCTION

This chapter describes the first stock enhancement trial of *Scylla* spp. in the Philippines. It aims to compare survival and growth of wild-released and hatchery-reared crabs in terms of species, sources and sizes classes. The effect of nursery conditioning of hatchery-reared crabs is investigated. Yield and catch per unit effort of wild crab catches only and catches including the stocked crabs are compared. Natural and fishing mortalities between species are also analysed. The appropriate species and size of crabs for release in this particular area is identified. Data obtained from this study will serve as baseline information for future enhancement programmes.

Overfishing of *Scylla* spp. has been observed at varying levels in different countries, as both national and international markets have developed, with resulting decreases in both size and abundance of mud crabs in many fisheries (Angell 1992; Overton et al. 1997; Kosuge 2001). In the late 1990s, as interest in aquaculture of mud crabs was increasing throughout Asia, commercial-scale hatchery production of *S. serrata* was not yet economically viable (Mann et al. 1999), so that all forms of mud crab culture depended on collection of natural seed (Overton et al. 1997). The lack of technology for the hatchery production of seed has not only hindered the development of the mud crab grow-out industry but has meant that unregulated fishing in many areas targets all life stages from first recruits to mature females (Le Vay 2001). However, recent developments suggest that commercial hatcheries will be operational in the near future (Fushimi and Watanabe 1999; Mann et al. 1999; Quinitio et al. 2002; Quinitio and Parado-Estepa 2003; Wang et al. 2005) and the technology to produce juveniles from the hatchery may lessen the pressure on wild stocks by decreasing the demand on juveniles for use in aquaculture. However, as human populations continue to grow, particularly in coastal rural areas, the need to derive income from fishing is unlikely to be lessened by the increase in aquaculture production, which mainly benefits pond owners. Thus, harvesting of wild populations continues, especially by low income fishers who mainly depend on fishing for their livelihoods.

The development of technologies for hatchery production of mud crabs also opens up the possibility of improvement of small scale fisheries through release of hatchery-reared juveniles. Bell and Nash (2004) and Bell et al. (2005) differentiate between 'stock enhancement' as increasing productivity of operational fisheries by overcoming recruitment limitations, and 'restocking' as rebuilding spawning biomass of severely depleted populations to levels where the fishery can once again support regular harvest. Releases of hatchery-reared juveniles and translocations of adults are used for restocking initiatives while stock enhancement includes a broader range of intervention such as release of hatchery-reared juveniles, capture, culture and release of wild-caught juveniles, thinning and relocation of dense aggregations of spat, and provision of additional habitat to increase settlement success (Bell et al. 2005). In Leber et al. (2004), aside from defining 'stock enhancement' as stocking cultured organisms to replenish or increase abundance of wild stocks, the term 'sea ranching' was also defined as stocking for put-grow-and-take food fisheries.

Marine stock enhancement is not a new concept. The idea of stocking to enhance marine fisheries dates back to the latter part of the 1800s after the first salmon hatchery in the United States was established in the 1870s (Blankenship and Leber 1995). This was followed by the release of marine finfish such as Atlantic salmon, cod, haddock, pollock and flounder (Blaxter 2000; Leber 2004). In Europe, the earliest attempts of stock enhancement of fish include releases of plaice in Norway in 1882 and in Scotland in 1894 and plaice transplants in Baltic and North Sea in 1893 (Blaxter 2000). In 1889, newly-hatched eggs and newly-settled juveniles of the European lobster *Homarus gammarus* were released in southern Norway (Salvanes 2001). The first attempt at scallop enhancement occurred in 1900 in Norway but was discontinued and resumed on a commercial scale in Europe and Japan in 1970 (Salvanes 2001). Stock enhancement has been attempted in finfish such as Pacific salmon (Beamish and Noakes 2004), coastal cod *Gadus morhua* (Moksness 2004), red drum *Sciaenops ocellatus* (Bert et al. 2003), common snook *Centropomus undecimalis* (Brennan et al. 2005), red snapper *Lutjanus campechanus* (Blaylock et al. 2000), and the striped mullet *Mugil cephalus* (Leber et al. 1995; Leber et al. 1996). It also encompasses invertebrates such as giant clams *Tridacna* spp. (Mingoa-Licuanan and Gomez 2002), scallops *Patinopecten yessoensis* (Kitada 1999), abalone *Haliotis iris* (Schiel 1993), topshell *Trochus niloticus* (Crowe et al. 2002), queen conch

Strombus gigas (Ray et al. 1994), sea cucumber *Holothuria scabra* (Dance et al. 2003), sea urchins *Strongylocentrotus intermedius* (Agatsuma et al. 2003), shrimp *Penaeus orientalis* (Xu et al. 1997), and lobsters *Homarus gammarus* (Jensen et al. 1994). These has recently been of interest in stock enhancement of crabs such as *Callinectes sapidus* (Davis et al. 2005; Zmora et al. 2005), *Portunus trituberculatus* (Okamoto 2004), and the mud crab *S. tranquebarica* (Secor et al. 2002). Bell and Gervis (1999) have tabulated some of the countries and territories in the tropical Pacific actively involved in stock enhancement programmes on giant clams, sea cucumbers, penaeid shrimps and some gastropods. Bartley et al. (2004) presented a listing of the species used for marine stocking in developing countries. Salvanes (2001) listed the main species of finfish, crustaceans, shellfish and echinoderms which are bred in captivity for release in stock enhancement programmes in North America, Europe and Japan. Japan, being the world leader in the development of artificial propagation methods for marine fishes and crustaceans (Secor et al. 2002), has approximately 80 species being ranched or researched for eventual stocking (Fushimi 2001; Salvanes 2001).

According to Bell and Nash (2004), access to technology for producing and releasing juveniles is not a sufficient rationale to proceed with restocking or stock enhancement. Careful decisions need to be made about whether these interventions are likely to be cost-effective in improving productivity. Jorgensen (2005) stressed that intentional introduction of a species for stock enhancement may have unforeseen effects. Bell and Gervis (1999) elaborated the possible risks of stock enhancement to the wild population such as introduction of diseases, modification of gene pools and increased inter- or intraspecific interactions. Thus, to ensure successful use of the stock enhancement concept and avoid repeating past mistakes, Blankenship and Leber (1995) proposed ten components of a so-called "responsible approach to marine stock enhancement" which embrace logical and conscientious strategies for applying aquaculture technology to help conserve and expand natural resources. These components include the need to 1) prioritize and select target species for enhancement, 2) develop a species management plan that identifies harvest opportunity, stock rebuilding goals, and genetic objectives, 3) define quantitative measures of success, 4) use genetic resource management to avoid deleterious genetic effects, 5) use disease and health management, 6) consider ecological, biological and life history

patterns when forming enhancement objectives and tactics, 7) identify release hatchery fish and assess stocking effects, 8) use an empirical process for defining optimum release strategies, 9) identify economic and policy guidelines, and 10) use adaptive management. Thus, stock enhancement can be considered a multidisciplinary technique that needs the involvement of biologists, geneticists, pathologists, statisticians, socio-economists, fisheries managers and the fishers. It is not just a simple "produce and release" technique, since many factors should be taken into consideration. Problems in the past that restricted the development of marine stock enhancement include lack of capability to evaluate success of hatchery releases, inability to culture marine fish beyond the early larval stages to the juvenile stage (Blankenship and Leber 1995) and the relative lack of fitness of reared fish released in the wild (Blaxter 2000). Recent advances in tagging technology, success in hatchery production for different species of marine organisms and conditioning strategies employed prior to releasing hatchery-reared organisms in the wild have overcome some of these problems in some cases.

Although stock enhancement has been primarily practised in developed countries, it has been considered lately in developing countries given the importance of fishery products to their livelihoods and economies (Bartley et al. 2004). In the Philippines, for example, where most of the fisheries resources have been overexploited, stock enhancement is being considered as possible strategy to save some sectors of the dwindling fisheries industry. Stock enhancement of some species such as the giant clam *Tridacna* spp. (Mingoa-Licuanan and Gomez 2002), abalone *Haliotis asinina* (W.G. Gallardo pers. comm.) and sea horse *Hippocampus kuda* and *H. barbouri* (L.M.B. Garcia pers. comm.) have been tested in the past. The present study is the first trial of release of the three species of *Scylla*.

The declining population of *Scylla* spp. in the selected study area, as observed by the local crab collectors who have been into this livelihood for decades, has been supported by the results of the present work reported in Chapters 3 and 4. The low yield, catch per unit effort, population density and the decreasing mean carapace width (CW) of crabs caught for the previous 43 months are evidence of an exploited mud crab population. Moreover, addition of new recruits may be inadequate to replace crabs lost to natural and increasing fishing mortality as recruitment to the

study area may be limited by the isolated nature of the habitat, having limited access to the open sea, only via the narrow mouth of the Naisud River. Taking all these factors into consideration, the study area was considered highly suitable for stock enhancement trials. The aim of the study was to determine the survival and growth of wild-released and hatchery-reared crabs, particularly examining the effect of nursery conditioning, size-at-release and species differences. Yield and catch per unit effort will be compared between the wild population and additional production including the stocked crabs. Survival, reflected by the percentage of crabs recovered, and growth rates between species (*S. olivacea*, *S. serrata*, *S. tranquebarica*), sources (wild-released; hatchery-reared, pond-conditioned; hatchery-reared, unconditioned), and sizes (20-79.9 mm, at 4.9 mm intervals) will be compared to have a basis for future enhancement programmes. From the results of this present work, the appropriate species and size of crabs for release in this particular area may be identified.

MATERIALS AND METHODS

Study Site

The study area was described in detail in Chapter 2 and is the same mangrove area where population and fisheries (Chapter 3) and population estimate and growth (Chapter 4) studies on wild *Scylla* spp. were conducted. Three release areas were chosen at the upper reaches of the three main branches of Naisud River. The mangrove area surrounding the river was divided into six collection areas, as described in Chapter 4.

Release sites located on each branch of Naisud River were identified with letters (A, B, C) from west to east, while collection areas were identified with numbers (1-6) (Fig. 5.1). Release Site A is at the tip of collection Area 1, which covers the whole length of the 600-m westernmost creek and is closest to the ponds and the road. Release Site B is the tip of collection Area 3 which encompasses the whole major branch of the main creek, 600 m from the main creek junction. It is also close to collection Area 4 which covers 450 m from the junction of Area 3 towards the tip of

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Figure 5.1. Release sites (top) and collection areas for monitoring recoveries (bottom) of tagged *Scylla* spp. for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; May 2004-November 2005.

the main creek, and to collection Area 2 which is an 800-m stretch from the junction of Area 3 to the west going downstream along the main creek. Release Site C is at the tip of collection Area 4, located at the upper reaches of the main creek. Collection Area 4 covers the whole 450 m of the main creek to the junction of Area 3. Area 5 covers the area where all three creeks converge and Area 6 is the 250-m stretch after Area 5 reaching to the seaward limit of the mangroves before the mouth of Naisud River.

The study was conducted from May 2004 to November 2005. Crab releases were completed during May 2004-September 2005 and monitoring of recoveries continued from June 2004 to November 2005. Release of tagged crabs was conducted in batches depending on the availability from the source and was done the day after the last day of routine sampling to allow dispersal and avoid immediate capture.

Source of released crabs

Wild *S. olivacea* were obtained from the replanted mangroves in New Buswang, Kalibo, Aklan. The area has been described in detail in Walton et al. (2006a). *S. olivacea*, measuring 30-79.9 mm CW, caught during spring tide within the duration of the experiment were purchased at PhP5 apiece (PhP100=£1.08, exchange rate from http://www.xe.com/ucc/convert.cgi as of 31 January 2006). They were maintained in plastic containers (20 cm high x 60 cm diameter) for up to 4 d, sprinkled with seawater enough to keep them moist and cool and fed chopped fish by-catch once daily. The water that accumulated in the container was poured out 4-6 hours after feeding to get rid of excess feeds. Crabs were again sprinkled with fresh seawater to provide them with a clean environment until the next feeding the following day. They were tagged on the last day of the sampling.

Hatchery-reared (HR) *S. olivacea*, *S. serrata* and *S. tranquebarica* were obtained from the Crustacean Hatchery, Tigbauan Main Station of the Southeast Asian Fisheries Development Center Aquaculture Department in Tigbauan, Iloilo, Philippines. The methods for seed production are described in detail by Quinitio et al. (2002) and Quinitio and Parado-Estepa (2003). Hatchery-reared crabs were either released directly from the hatchery to the mangroves without undergoing conditioning in

ponds (HR-unconditioned) or first transferred to the earthen ponds in Dumangas Brackishwater Station (Fig. 5.2) and reared for 1-1.5 months prior to release (HR-conditioned). HR-unconditioned crabs were reared individually in perforated, plastic containers (round: 5 cm high x 15 cm diameter; or rectangular: 16 x 10 x 5 cm) placed in concrete tanks (1 m high x 4 m diameter) provided with filtered seawater (salinity 32-34 p.s.u., temperature 26-30.5°C) and aeration. Water was maintained at 40-50 cm depth and replaced at 30-50% of the volume every 2-3 days depending on water quality. They were fed *ad libitum* twice daily with minced fish by-catch, mussel or small shrimps *Acetes* spp. and excess feeds were siphoned out prior to water change. Upon reaching 20 mm CW in the hatchery, crabs for conditioning were transferred to brackish water earthen ponds in Figure 5.2 and reared at a density of 1-2 crabs m^{-2}. They were fed twice daily with minced fish by-catch and mussel placed in feeding trays. Pond water was changed every spring tide. However, addition of water was done 1-2 times weekly to replace water loss due to seepage and evaporation and to maintain water depth at 80-100 cm. At a minimum size of 30 mm CW, both HR-unconditioned and HR-conditioned crabs were tagged for release, except for the first batch of HR-unconditioned *S. serrata* (Batch 1) and the HR-unconditioned *S. tranquebarica* (Batch 5) where the minimum size was 20 mm CW.

Tagging and release

Unlike in Chapter 4, crabs in the present study were tagged in batches. Crabs were measured, sorted and grouped into size classes from 20.0 to 79.9 mm CW at 4.9 mm intervals and placed in containers; sexually mature females were excluded. Tagging started from the smallest size class. A reference tag was taken before tagging commenced and at the end of each size class thereafter. The reference tags separated the crabs into different batches and different size classes within each batch. Each crab was tagged at the base of its third walking leg on the left side to separate them from the crabs used for mark-recapture study in Chapter 4, which were tagged on the right. All the other tagging details follow the same protocols used in Chapter 4.

For wild *S. olivacea*, when all crabs had been tagged, they were temporarily placed in shallow basins provided with a little seawater and covered with mangrove twigs and leaves. They were then transported the following day to the release site in bags made

Figure 5.2. Earthen ponds with net enclosures in the Dumangas Brackishwater Station of the Southeast Asian Fisheries Development Center Aquaculture Department, Dumangas, Iloilo, Philippines where hatchery-produced *Scylla* spp. were conditioned for 1-1.5 months prior to release in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, November 2004-September 2005.

from dried palm leaves. Distance from the source in Kalibo, Aklan where the crabs were tagged to the release site in Ibajay, Aklan is approximately 35 km.

For hatchery-produced crabs, tagging was done the day before release and conducted where the crabs were being reared. This was in the hatchery for HR-unconditioned crabs (Tigbauan, Iloilo) and at the pond-side for HR-conditioned ones (Dumangas, Iloilo). When all crabs had been tagged, they were placed in containers and transported the following day to the release site, approximately 210 km from the hatchery and 190 km from the ponds. Crabs from both the hatchery and pond were transported at 0400 h for cooler conditions during the long journey. They were transported in rectangular, plastic containers (40 x 30 x 10 cm) lined with transparent plastic and covered with mangrove twigs and leaves which served as shelter (Fig. 5.3). These containers were placed on top of each other and tied securely to keep the crabs from escaping. Despite transporting the crabs in air conditioned vehicles, during the trip, they were occasionally sprinkled with seawater to prevent mortality caused by desiccation.

Crabs were released into one of the three areas shown in Figure 5.1. A total of 30 batches of crabs were released overall: 16 batches (14 wild, 2 HR-conditioned) of *S. olivacea*, 12 batches (7 HR-unconditioned, 5 HR-conditioned) of *S. serrata*, and 2 batches (1 HR-unconditioned, 1 HR-conditioned) of *S. tranquebarica*. A summary of release data for each batch is shown in Table 5.1.

Monitoring of environmental conditions

Temperature and salinity were measured using a hand-held YSI conductivity meter. Measurements were taken at the release sites when crabs were released (see Table 5.1 for release dates). Temperature ranged from 26.11 (January 2005) to 30.83°C (August 2005), mean 28.25 ± 0.30°C; and salinity from 3.8 (September 2005) to 34.2 p.s.u. (May 2004), mean 21.20 ± 2.08 p.s.u..

Figure 5.3. Unconditioned or conditioned hatchery-produced *Scylla* spp. were transported from the hatchery or pond in plastic containers lined with plastic and covered with mangrove leaves and released in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; May 2004-September 2005.

Table 5.1. Data summary of *Scylla* spp. released in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, May 2004-September 2005 showing ranges of tag codes used per batch, number of crabs per batch (N), date and site of release. Wild-released crabs are referred to as wild, hatchery-produced, unconditioned crabs as hatchery and hatchery-produced, pond-conditioned crabs as pond.

Species/ Batch	Source	Code	N	Date	Site
Scylla olivacea					
2	Wild	4026-4221	110	23 May 2004	B
3	Wild	4225-4550	178	8 June 2004	A
4	Wild	4557-4836	157	23 June 2004	B
7	Wild	6297-6508	113	22 July 2004	A
9	Wild	7153-7276	63	6 August 2004	B
10	Wild	7278-7395	63	20 August 2004	A
12	Wild	8068-8118	25	2 September 2004	B
14	Wild	8688-8774	44	17 September 2004	A
20	Wild	11036-11191	80	17 December 2004	B
24	Wild	12350-12653	167	26 February 2005	C
25	Wild	12656-12918	143	13 March 2005	B
26	Wild	12920-13277	203	2 April 2005	C
27	Wild	13279-13695	242	15 April 2005	B
28	Wild	13698-14048	201	2 May 2005	C
29	Pond	14056-14433	191	16 August 2005	C
30	Pond	14435-14812	255	17 September 2005	B
Scylla serrata					
1	Hatchery	2908-3287	175	4 May 2004	C
6	Hatchery	5215-5840	327	8 July 2004	A
8	Hatchery	6513-7145	221	6 August 2004	C
11	Hatchery	7398-8065	356	2 September 2004	A
13	Hatchery	8122-8681	282	17 September 2004	C
15	Hatchery	8777-9122	187	20 November 2004	C
18	Hatchery	10313-10821	284	17 December 2004	B
16	Pond	9124-10123	588	20 November 2004	C
17	Pond	10124-10312	102	20 November 2004	C
19	Pond	10823-11034	102	17 December 2004	C
21	Pond	11195-11385	104	14 January 2005	C
23	Pond	11809-12347	310	30 January 2005	B
Scylla tranquebarica					
5	Hatchery	5030-5213	94	8 July 2004	C
22	Pond	11385-11805	240	14 January 2005	B

Monitoring of recoveries

Monitoring of recoveries in fishers' catches commenced in June 2004, one month after the first batch of HR-unconditioned *S. serrata* and wild *S. olivacea* had been released. To encouraged unbiased return of tagged animals, all *Scylla* spp. caught on the succeeding spring tides were purchased at PhP100 kg^{-1}, regardless of species, sex and size and checked for the presence of tagged animals. Tagged crabs were separated from the untagged ones, segregated by source or collection area. For each individual, the species was identified, sexual maturation determined and CW and body weight (BW) measured. The third walking leg on the left of each tagged crab was then removed and the tag dissected from its muscle tissue. The tags were kept and read following procedures described in detail in Chapter 4.

Data analysis

Data were analysed from June 2004, the start of monitoring for recoveries from stock enhancement trials to November 2005, the termination of the study. However, for comparison purposes, background fisheries data from April 2002 are also presented. Data were analysed statistically and graphs were plotted using Minitab version 14. For convenience, *Scylla* spp. catches from the wild population are referred to here as 'wild', those released animals as 'stocked', and combined catches from wild and released as 'total'. Among the sources of released crabs, wild *S. olivacea* are referred to as 'wild-released', hatchery-reared, pond-conditioned *Scylla* spp. as 'HR-conditioned' and hatchery-reared, unconditioned crabs as 'HR-unconditioned'.

Yield and CPUE

Yield was calculated for landings from wild crabs, stocked crabs, and total catches (wild + stocked) for all *Scylla* spp. Catch per unit effort (CPUE) was calculated both for number (crabs $gear^{-1}$ d^{-1}) and biomass (g $gear^{-1}$ d^{-1}). Data from wild catches starting April 2002 to May 2004 were compared with the wild catches from June 2004 to November 2005. Wild catches from June 2004 to November 2005 were compared with total catches from the same period. A two-sample t-test was used to compare both wild catches from different months (April 2002-May 2004 vs. June 2004-

November 2005) and paired t-test to compare wild and total catches (wild + stocked) from June 2004 to November 2005.

Growth

Growth rates of all *Scylla* spp. from different sources were calculated and compared using regression analysis of individual growth (mm CW at recovery – mm CW at release) plotted against time or days (interval between release and recovery) following the general linear regression model:

$$y = a + bx; \qquad \text{(Equation 1)}$$

where: y = dependent variable; final CW – initial CW in mm

a = intercept; value of y at $x = 0$

b = slope of the line; the change in y for every unit change in x

x = independent variable; day of recovery – day of release in days

Weight was not used in the analysis of growth rates because weight data were obtained only during recovery. Prior to tagging, crabs were not individually weighed but sorted and grouped by CW size class.

Recovery of released crabs and Mortality

The number of crabs recovered from different batches released was noted every spring tide and compared between species, sources, size classes and release sites. Mortality (Z) rates were calculated following Gulland (1983) and King (1995) for batches where at least 100 crabs were released and more than 30% of them recovered. Mortality rates were calculated for each species from pooled values of the total number of tagged crabs released and the number of crabs recovered starting one month after release and the succeeding month thereafter until the last crab was recovered. Regression analysis of the natural log (log_e) of the pooled monthly recoveries plotted against the midpoint of the corresponding time interval (month) was used following the general linear regression model:

$$y = a + bx;$$ (Equation 2)

where: y = dependent variable; natural log (log_e) of monthly recoveries (pooled)

a = intercept; value of y at $x = 0$

b = slope of the line; the change in y for every unit change in x

x = independent variable; mid-point of the corresponding time interval (month)

The natural log of monthly recoveries plotted against the mid-point of the time interval (month) will give the value for the instantaneous rate of total mortality, Z which is -b. The intercept on the y-axis (a) can be used to estimate fishing mortality (F), as follows:

$$a = \ln\frac{N_0 F}{Z} + \ln(1 - \exp^{(-Zt)});$$ (Equation 3)

where: a = intercept; value of y at $x = 0$

N_0 = number of tagged crabs released (pooled)

F = fishing mortality

Z = total mortality, -b

t = time interval between recoveries, $t = 1$ month

therefore:

$$F = \frac{Z \exp^{a}}{N_0(1 - \exp^{(-Zt)})};$$ (Equation 4)

From the values of total mortality (Z) and fishing mortality (F), natural mortality (M) was computed as follows:

$$M = Z - F;$$ (Equation 5)

Movement

It was not possible to quantify crab movement in terms of actual distance travelled by individual animals from the day of release to the day of recovery. However, dispersal of crabs from each release site to the six collection areas was recorded. Distances were estimated from points and track lines obtained during mapping using the GPS Mapping Software OziExplorer version 3.90.3a.

RESULTS

Yield

Figure 5.4 shows the total monthly catch of wild *Scylla* spp. from April 2002 to May 2004 and the total catch from June 2004 to November 2005. The average monthly yield of wild catches for the period April 2002-November 2005 was 17.0 ± 1.32 kg. The average monthly yield of mud crab from the wild population from April 2002 to May 2004 was 17.86 ± 1.3 kg while from June 2004 to November 2005 was 15.80 ± 1.3 kg. Both means do not significantly differ from each other (T-value=0.71, $p>0.05$). However, average monthly yield from wild catch (June 2004-November 2005) was significantly lower than the average monthly yield of the total catch in the same period (23.1 ± 3.1 kg; T-value=-6.15, $p<0.001$) giving an overall increase in yield by 46%. Monthly yield per hectare (kg ha^{-1}) was obtained by dividing the total monthly catch by a 50-ha mangrove area, multiplying this value by 12 gives the annual yield per hectare (kg ha^{-1} yr^{-1}). Comparing crab catches for the last 18 months of the sampling, when landings consisted of crabs coming from the wild and stocked animals, wild catches give a yield of 0.32 kg ha^{-1} or 3.79 kg ha^{-1} yr^{-1} while total catches give a yield of 0.46 kg ha^{-1} or 5.54 kg ha^{-1} yr^{-1}. The decrease in yield starting May 2005 is caused by the reduction of the sampling period from 8 d $month^{-1}$ (two spring tides) to just 4 d $month^{-1}$ (one spring tide).

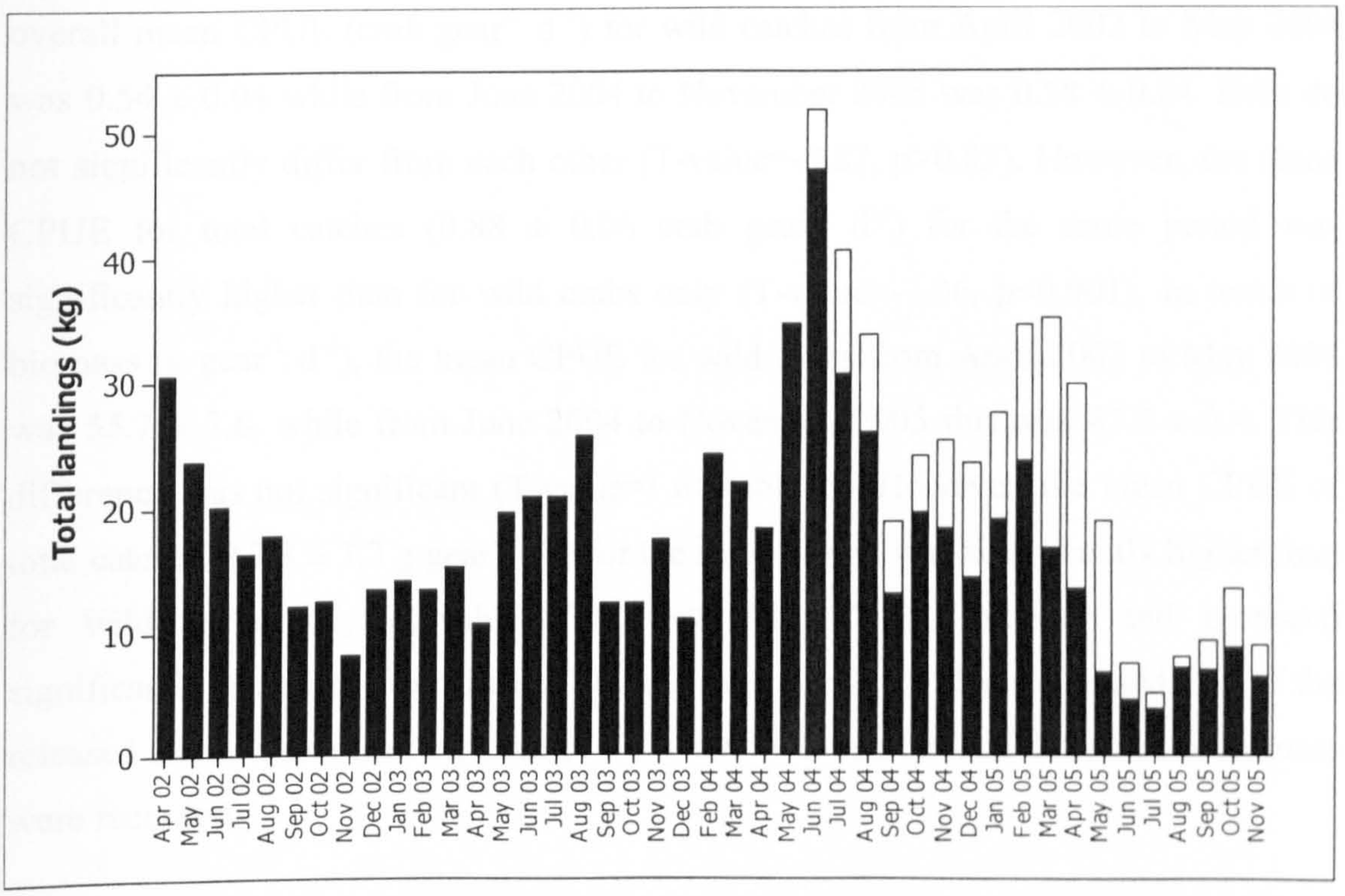


Figure 5.4. Monthly catches (kg) of *Scylla* spp. from the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, April 2002-November 2005. Black bars represent monthly catches of the wild crabs, white bars represent monthly catches of stocked *Scylla* spp.

CPUE

Figure 5.5 shows the monthly CPUE both for number (crabs gear^{-1} d^{-1}) and biomass (g gear^{-1} d^{-1}) of wild (April 2002-November 2005), stocked and total catches (June 2004-November 2005). Tables 5.2 and 5.3 show the monthly CPUE ± S.E. for number and biomass, respectively, of wild, stocked and total *Scylla* spp. catches. The overall mean CPUE for wild crabs for the period April 2002-November 2005 was 0.56 ± 0.03 crab gear^{-1} d^{-1} for number and 52.4 ± 2.60 g gear^{-1} d^{-1} for biomass. The overall mean CPUE (crab gear^{-1} d^{-1}) for wild catches from April 2002 to May 2004 was 0.54 ± 0.04 while from June 2004 to November 2005 was 0.58 ± 0.04. Both do not significantly differ from each other (T-value=-0.87, $p>0.05$). However, the mean CPUE for total catches (0.88 ± 0.06 crab gear^{-1} d^{-1}) for the same period was significantly higher than for wild crabs only (T-value=-7.06, $p<0.001$). In terms of biomass (g gear^{-1} d^{-1}), the mean CPUE for wild crabs from April 2002 to May 2004 was 55.7 ± 3.6, while from June 2004 to November 2005 this was 47.7 ± 3.4. This difference was not significant (T-value=1.61, $p>0.05$). However, the mean CPUE of total catches (67.8 ± 3.3 g gear^{-1} d^{-1}) for the same period was significantly higher than for wild crabs only (T-value=-7.46, $p<0.001$). CPUE (number and biomass) significantly increased from June 2004 to November 2005, due to the addition of the released crabs. An overall increase of 51% in CPUE number and 42% CPUE biomass were recorded.

Growth

Growth here refers to the difference between final CW and initial CW ($CW_F - CW_I>0$) between periods of recovery and release. Combining all three species of *Scylla* recovered regardless of source, the results of the general linear regression model showed that crabs grew at a mean rate of 0.18 mm d^{-1}. Using the same model for different species, regardless of source, showed that *S. olivacea* grew at a mean rate of 0.27 mm d^{-1}, *S. serrata* 0.13 mm d^{-1}, and *S. tranquebarica* 0.18 mm d^{-1}.

Analysis of variance was used to compare growth rates between the three *Scylla* spp. from different sources (wild-released and HR-conditioned *S. olivacea*, HR-unconditioned and HR-conditioned *S. serrata* and HR-conditioned *S. tranquebarica*).

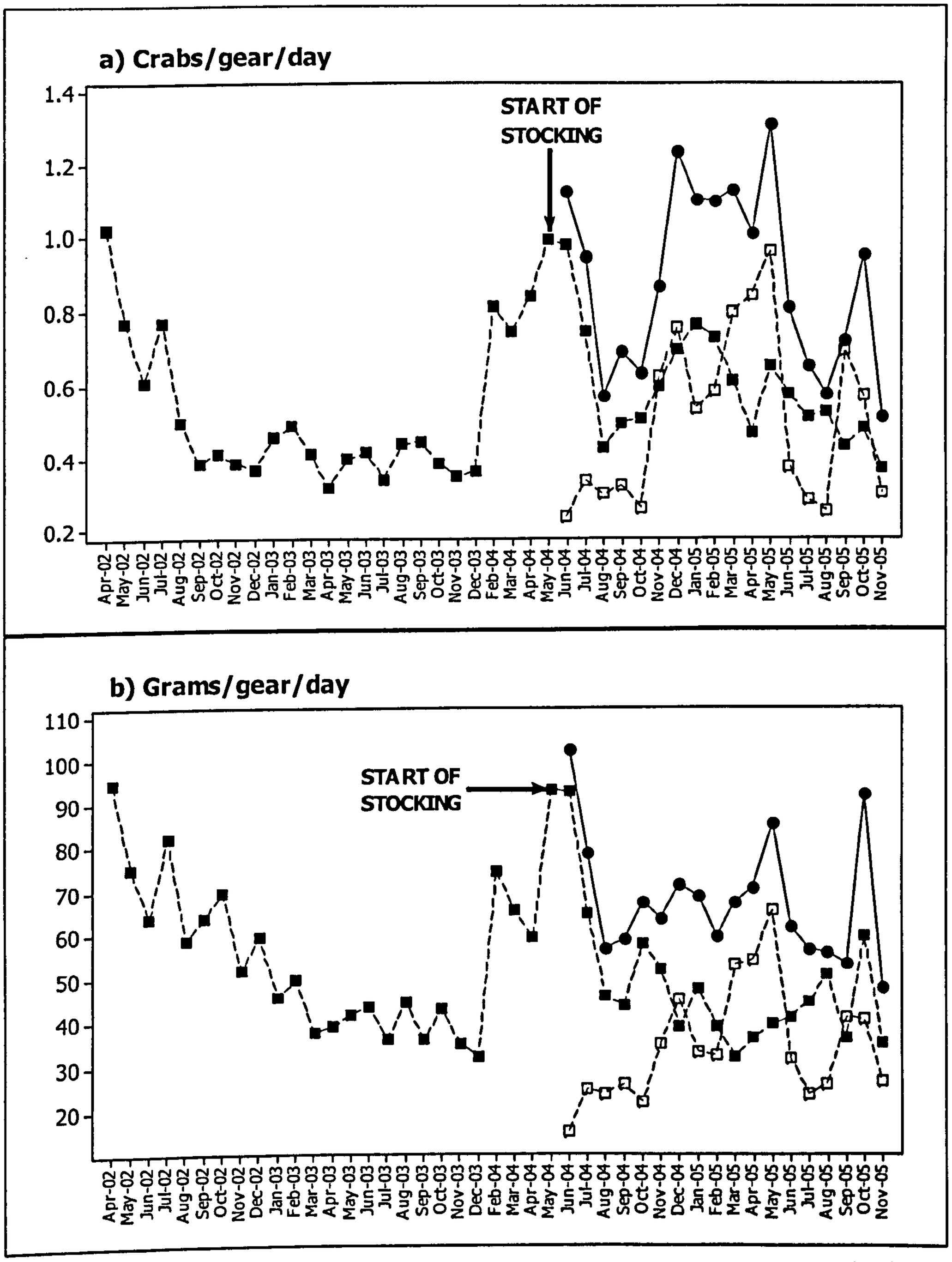


Figure 5.5. Monthly catch per unit effort (CPUE) expressed as a) crabs $gear^{-1}$ d^{-1} b) g $gear^{-1}$ d^{-1} of *Scylla* spp. caught from the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, April 2002-November 2005. Filled circles = total catches of *Scylla* spp., filled squares = catches of wild *Scylla* spp., empty squares = catches of stocked *Scylla* spp.

Table 5.2. Monthly mean CPUE ± S.E. (crab gear^{-1} d^{-1}) of wild, stock and total catches of *Scylla* spp. from the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, April 2002-November 2005.

Month	Wild	Stock	Total
April **2002**	1.01 ± 0.16		1.01 ± 0.16
May 2002	0.77 ± 0.06		0.77 ± 0.06
June 2002	0.61 ± 0.06		0.61 ± 0.06
July 2002	0.77 ± 0.12		0.77 ± 0.12
August 2002	0.50 ± 0.05		0.50 ± 0.05
September 2002	0.39 ± 0.04		0.39 ± 0.04
October 2002	0.41 ± 0.05		0.41 ± 0.05
November 2002	0.39 ± 0.07		0.39 ± 0.07
December 2002	0.37 ± 0.05		0.37 ± 0.05
January **2003**	0.45 ± 0.06		0.45 ± 0.06
February 2003	0.49 ± 0.10		0.49 ± 0.10
March 2003	0.41 ± 0.07		0.41 ± 0.07
April 2003	0.32 ± 0.02		0.32 ± 0.02
May 2003	0.40 ± 0.03		0.40 ± 0.03
June 2003	0.41 ± 0.02		0.41 ± 0.02
July 2003	0.34 ± 0.02		0.34 ± 0.02
August 2003	0.44 ± 0.03		0.44 ± 0.03
September 2003	0.44 ± 0.03		0.44 ± 0.03
October 2003	0.38 ± 0.02		0.38 ± 0.02
November 2003	0.35 ± 0.03		0.35 ± 0.03
December 2003	0.36 ± 0.02		0.36 ± 0.02
February **2004**	0.81 ± 0.09		0.81 ± 0.09
March 2004	0.74 ± 0.07		0.74 ± 0.07
April 2004	0.84 ± 0.07		0.84 ± 0.07
May 2004	0.99 ± 0.08		0.99 ± 0.08
June 2004	0.98 ± 0.08	0.24 ± 0.04	1.12 ± 0.10
July 2004	0.74 ± 0.06	0.33 ± 0.06	0.94 ± 0.08
August 2004	0.42 ± 0.03	0.30 ± 0.04	0.57 ± 0.04
September 2004	0.49 ± 0.09	0.32 ± 0.05	0.69 ± 0.11
October 2004	0.50 ± 0.05	0.26 ± 0.04	0.63 ± 0.06
November 2004	0.60 ± 0.07	0.62 ± 0.13	0.86 ± 0.09
December 2004	0.69 ± 0.09	0.75 ± 0.10	1.23 ± 0.16
January **2005**	0.76 ± 0.08	0.53 ± 0.09	1.10 ± 0.13
February 2005	0.72 ± 0.07	0.58 ± 0.08	1.09 ± 0.10
March 2005	0.61 ± 0.06	0.79 ± 0.07	1.12 ± 0.10
April 2005	0.47 ± 0.05	0.84 ± 0.08	1.00 ± 0.10
May 2005	0.65 ± 0.08	0.96 ± 0.15	1.31 ± 0.15
June 2005	0.57 ± 0.11	0.37 ± 0.09	0.81 ± 0.14
July 2005	0.51 ± 0.08	0.28 ± 0.06	0.65 ± 0.09
August 2005	0.52 ± 0.10	0.25 ± 0.12	0.57 ± 0.09
September 2005	0.43 ± 0.07	0.69 ± 0.19	0.72 ± 0.17
October 2005	0.48 ± 0.07	0.57 ± 0.13	0.95 ± 0.14
November 2005	0.37 ± 0.05	0.30 ± 0.09	0.51 ± 0.08

Table 5.3. Monthly mean CPUE ± S.E. (g gear^{-1} d^{-1}) of wild, stock and total catches of *Scylla* spp. from the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, April 2002-November 2005.

Month	Wild	Stock	Total
April **2002**	94.5 ± 17.7		94.5 ± 17.7
May 2002	75.2 ± 7.4		75.2 ± 7.4
June 2002	63.9 ± 8.8		63.8 ± 8.8
July 2002	81.8 ± 14.3		81.8 ± 4.3
August 2002	58.5 ± 5.5		58.5 ± 5.5
September 2002	63.8 ± 10.3		63.8 ± 10.3
October 2002	69.7 ± 9.3		69.7 ± 9.3
November 2002	51.8 ± 9.0		51.8 ± 9.0
December 2002	59.3 ± 10.2		59.3 ± 10.2
January **2003**	45.6 ± 6.3		45.6 ± 6.3
February 2003	49.4 ± 9.6		49.4 ± 9.6
March 2003	37.4 ± 7.7		37.4 ± 7.7
April 2003	38.8 ± 6.2		38.8 ± 6.2
May 2003	41.6 ± 3.4		41.6 ± 3.4
June 2003	43.3 ± 3.9		43.3 ± 3.9
July 2003	36.0 ± 2.2		36.0 ± 2.2
August 2003	44.5 ± 3.6		44.5 ± 3.6
September 2003	35.9 ± 2.0		35.9 ± 2.0
October 2003	43.0 ± 4.9		43.0 ± 4.9
November 2003	35.1 ± 3.2		35.1 ± 3.2
December 2003	32.0 ± 3.0		32.0 ± 3.0
February **2004**	74.4 ± 8.2		74.4 ± 8.2
March 2004	65.4 ± 6.8		65.4 ± 6.8
April 2004	59.3 ± 5.3		59.3 ± 5.3
May 2004	93.2 ± 9.6		93.2 ± 9.6
June 2004	92.8 ± 9.1	15.4 ± 2.5	102.1 ± 10.1
July 2004	64.9 ± 5.3	24.8 ± 3.6	78.6 ± 6.1
August 2004	46.0 ± 3.6	23.8 ± 3.4	56.8 ± 3.8
September 2004	43.6 ± 4.7	25.9 ± 2.9	59.0 ± 5.2
October 2004	58.0 ± 11.0	21.8 ± 4.5	67.5 ± 10.6
November 2004	52.1 ± 7.6	35.0 ± 6.2	63.9 ± 6.5
December 2004	38.9 ± 5.2	45.2 ± 5.8	71.5 ± 7.6
January **2005**	47.7 ± 6.8	33.2 ± 5.3	68.9 ± 8.5
February 2005	39.0 ± 4.3	32.4 ± 4.4	59.6 ± 5.4
March 2005	32.1 ± 2.5	53.0 ± 4.3	67.3 ± 4.7
April 2005	36.3 ± 4.8	54.4 ± 4.9	70.7 ± 6.5
May 2005	40.0 ± 4.3	65.5 ± 10.2	85.6 ± 7.9
June 2005	41.2 ± 8.2	31.8 ± 6.1	61.8 ± 11.0
July 2005	44.9 ± 5.6	23.5 ± 5.8	56.7 ± 6.4
August 2005	50.9 ± 9.0	26.0 ± 16.8	56.1 ± 8.5
September 2005	36.2 ± 5.1	41.3 ± 11.1	53.6 ± 9.4
October 2005	59.8 ± 9.5	40.7 ± 8.9	92.1 ± 11.1
November 2005	35.3 ± 6.5	26.6 ± 7.9	48.1 ± 7.1

HR-unconditioned *S. tranquebarica* were not included in the analysis because only one of the 94 crabs released was recovered. Figure 5.6 shows the CW gain between release and recovery for the three *Scylla* spp. from different sources. Monthly growth rates from each species-source group from the highest were 11.7 mm month^{-1} for HR-conditioned *S. olivacea*, 7.6 mm month^{-1} for wild-released *S. olivacea*, 6 mm month^{-1} for HR-conditioned *S. tranquebarica*, 4.7 mm month^{-1} for HR-unconditioned *S. serrata* and 3.7 mm month^{-1} for HR-conditioned *S. serrata*. Table 5.4 shows the result of the regression analysis (slopes and intercepts of each species-source group compared), the analysis of variance table and the comparisons between species-source regression slopes and intercepts with the respective group averages. Of the five species-source groups compared, HR-conditioned *S. olivacea* exhibited the significantly highest growth rate (ANOVA, $F=13.05$, $p<0.001$). Growth rates of wild-released and HR-conditioned *S. olivacea* were significantly higher than the overall mean (ANOVA, $T=2.96$, $p<0.01$ and $T=7.13$, $p<0.001$, respectively), while HR-unconditioned and HR-conditioned *S. serrata* had significantly lower growth rates (ANOVA, $T=-6.34$, $p<0.001$ and $T=-10.91$, $p<0.001$, respectively). The growth rate of HR-conditioned *S. tranquebarica* showed no significant difference from the overall mean (ANOVA, $T=-1.40$, $p>0.05$).

Of the wild-released *S. olivacea* recovered, 29.3% showed no linear growth compared with 15.5% among the HR-conditioned ones. The maximum period between release and recovery of crabs showing no growth was 86 and 22 d, respectively. Of the HR-conditioned *S. serrata* recovered, 26.9% showed no linear growth compared with 7.4% among the HR-unconditioned ones. For these groups, the maximum period between release and recovery was 109 and 130 d, respectively. For *S. tranquebarica*, 24.3% of the recovered HR-conditioned crabs did not exhibit any linear growth, up to a maximum of 73 d from release.

Recovery of released crabs

The percentage of recovered *Scylla* spp. in total monthly catches ranged from 11.9% (June 2004) to 62.3% (May 2005), mean ± S.E.=34.8 ± 3.2% (Fig. 5.7). Recovery rates from all batches of *S. olivacea*, *S. serrata* and *S. tranquebarica* are shown in Figures 5.8, 5.9, and 5.10, respectively. Of the three *Scylla* spp. from different sources,

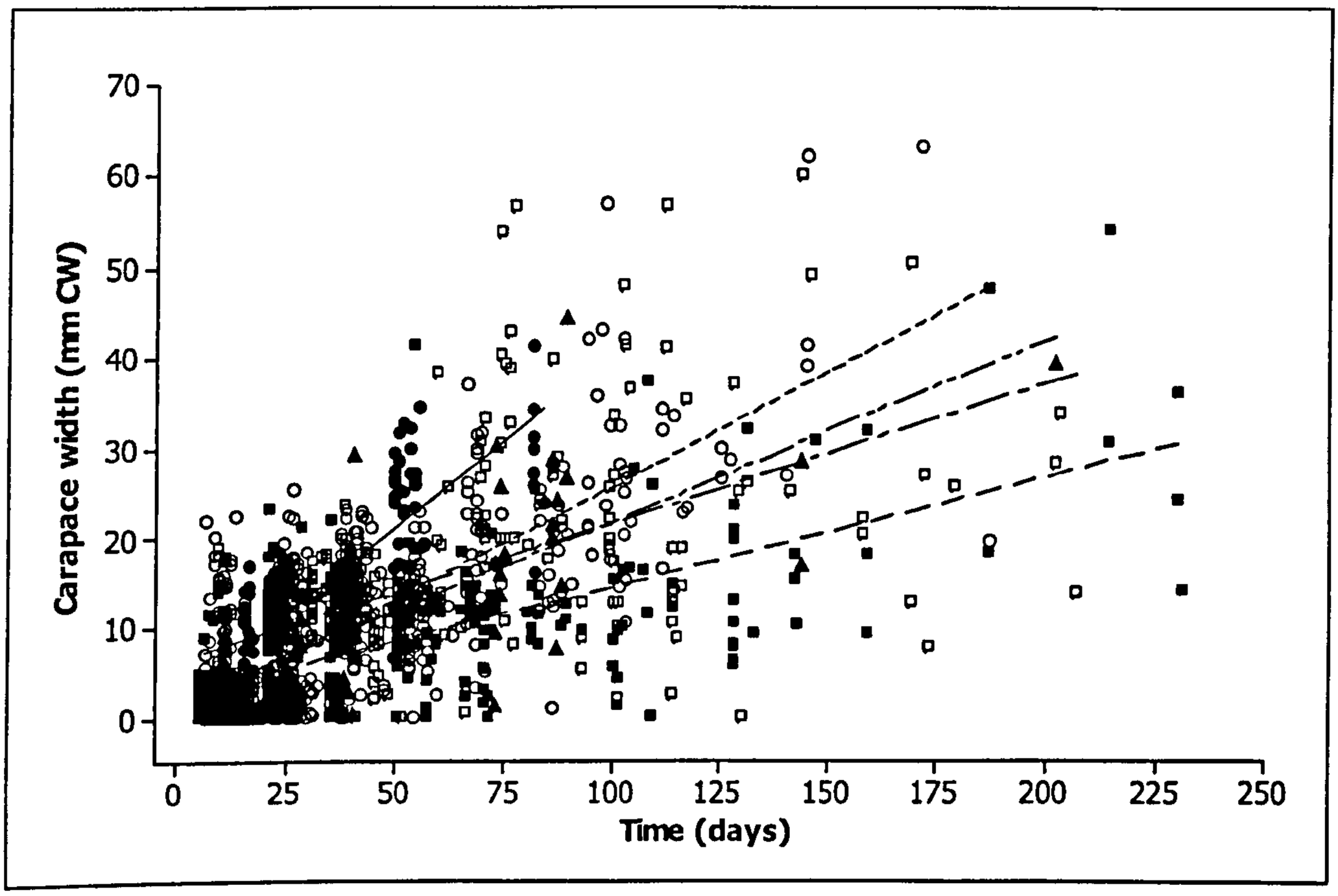


Figure 5.6. Comparison of growth rates (mm CW) between hatchery-produced, pond-conditioned and wild-released *Scylla olivacea*, hatchery-produced, pond-conditioned and hatchery-produced, unconditioned *Scylla serrata*, and hatchery-produced, pond-conditioned *Scylla tranquebarica* released in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, May 2004-September 2005. Hatchery-produced, pond-conditioned *S. olivacea* = filled circle and solid line; wild-released *S. olivacea* = empty circle and short broken line; hatchery-produced, pond-conditioned *S. serrata* = filled square and long broken line; hatchery-produced, unconditioned *S. serrata* = empty square and one long, one short broken line; hatchery-produced, pond-conditioned *S. tranquebarica* = filled triangle and one long, two short broken line.

Table 5.4. Analysis of carapace width (mm CW) increase between days of release and recovery (time) in hatchery-produced, pond-conditioned (SO-pond) and wild-released (SO-wild) *Scylla olivacea*, hatchery-produced, pond-conditioned (SS-pond) and hatchery-produced, unconditioned (SS-hatch) *Scylla serrata*, and hatchery-produced, pond-conditioned (ST-pond) *Scylla tranquebarica*.

a) Regression analysis of the increase in mm CW with time

Species-source	Slope ± S.E (mm d^{-1})	Intercept ± S.E (mm)
SO-pond	0.390 ± 0.03	1.856 ± 0.89
SO-wild	0.252 ± 0.01	0.627 ± 0.40
SS-pond	0.122 ± 0.01	2.397 ± 0.46
SS-hatch	0.155 ± 0.01	6.297 ± 0.70
ST-pond	0.201 ± 0.02	1.860 ± 0.81

b) Analysis of variance table with time in days as covariate; Seq=sequential, Adj=adjusted for entry order into the model

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Species-source	4	19553	1963	491	13.05	<0.001
Days	1	62963	33823	33823	899.29	<0.001
Species-source*Days	4	8020	8020	2005	53.31	<0.001
Error	1870	70331	70331	38		
Total	1879	160867				

c) Comparisons between species-source regression slopes and intercepts and their respective group averages. Slope differences in mm CW d^{-1}, intercept differences in mm. P values with * are significantly different from the average.

		Average	Difference	T-value	P
Species-source	Slopes	0.224 ± 0.01			
SO-pond			0.166	7.13	<0.001*
SO-wild			0.028	2.96	0.003*
SS-pond			-0.102	-10.91	<0.001*
SS-hatch			-0.069	-6.34	<0.001*
ST-pond			-0.023	-1.40	0.161
Species-source	Intercepts	2.61 ± 0.34			
SO-pond			-0.752	-0.85	0.397
SO-wild			-1.980	-4.96	<0.001*
SS-pond			-0.210	-0.46	0.647
SS-hatch			3.690	5.26	<0.001*
ST-pond			-0.747	-0.92	0.358

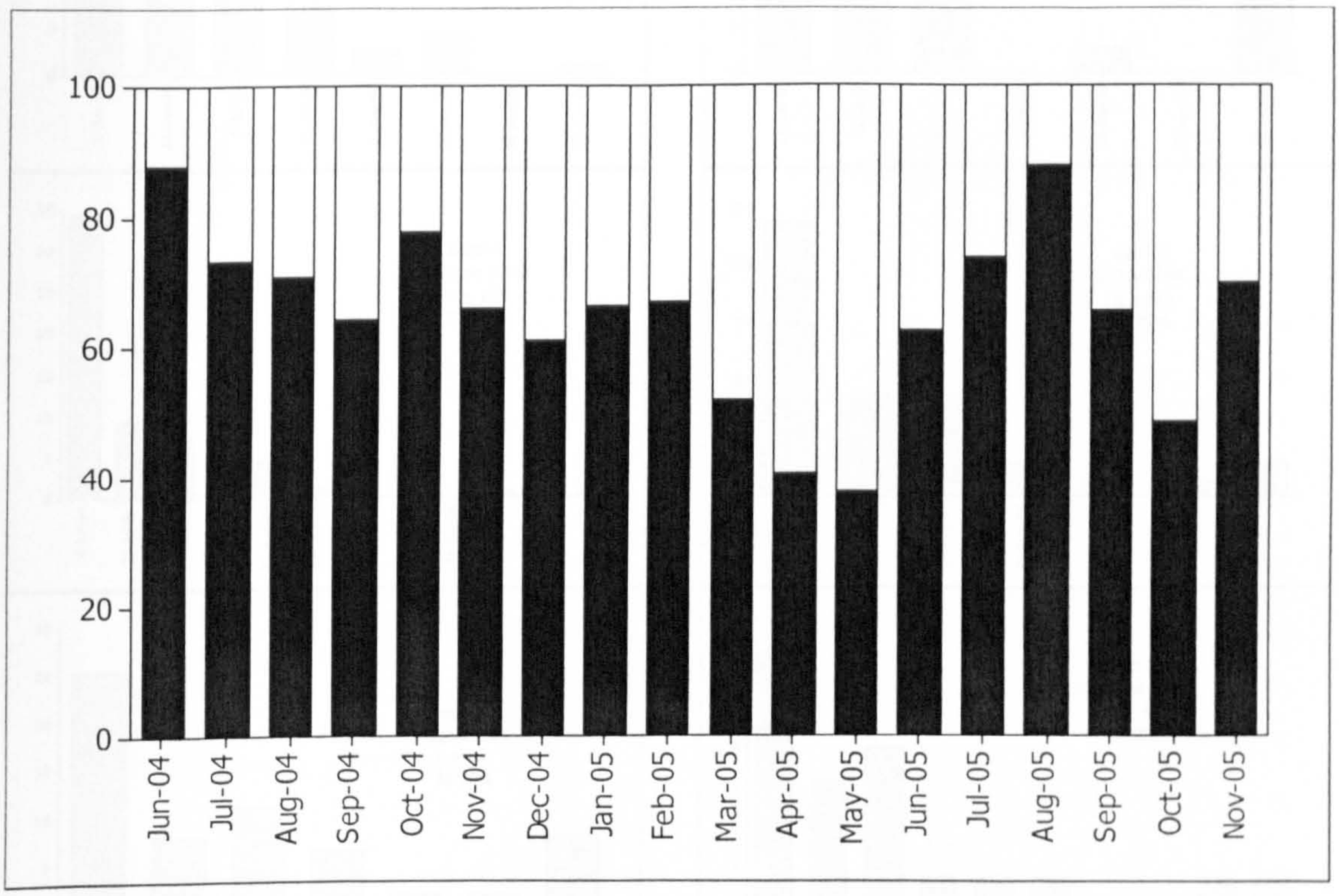
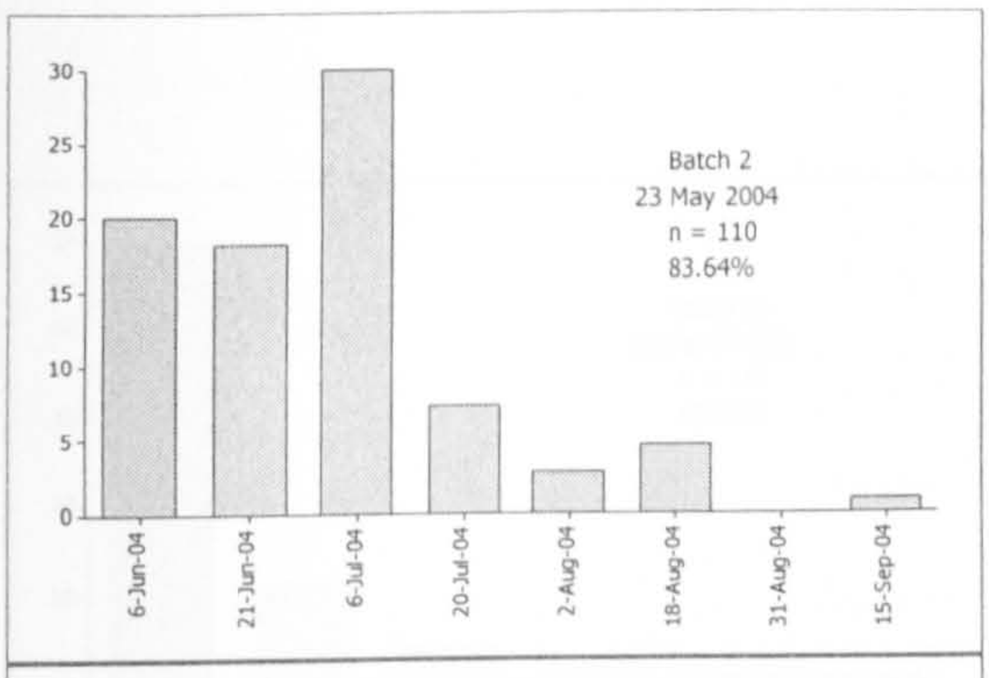
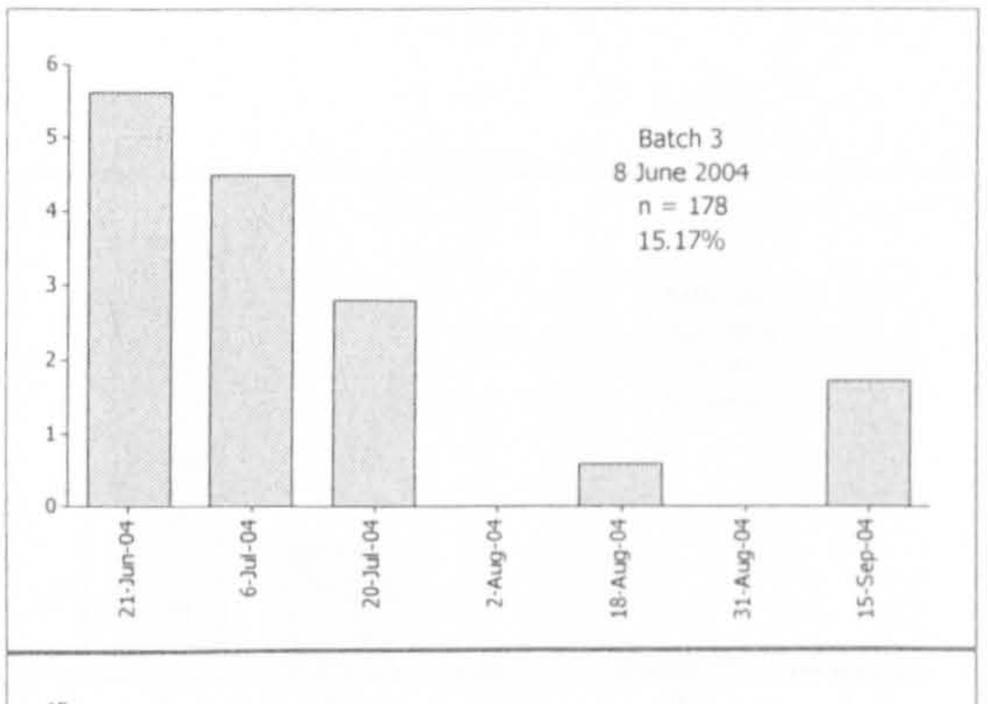
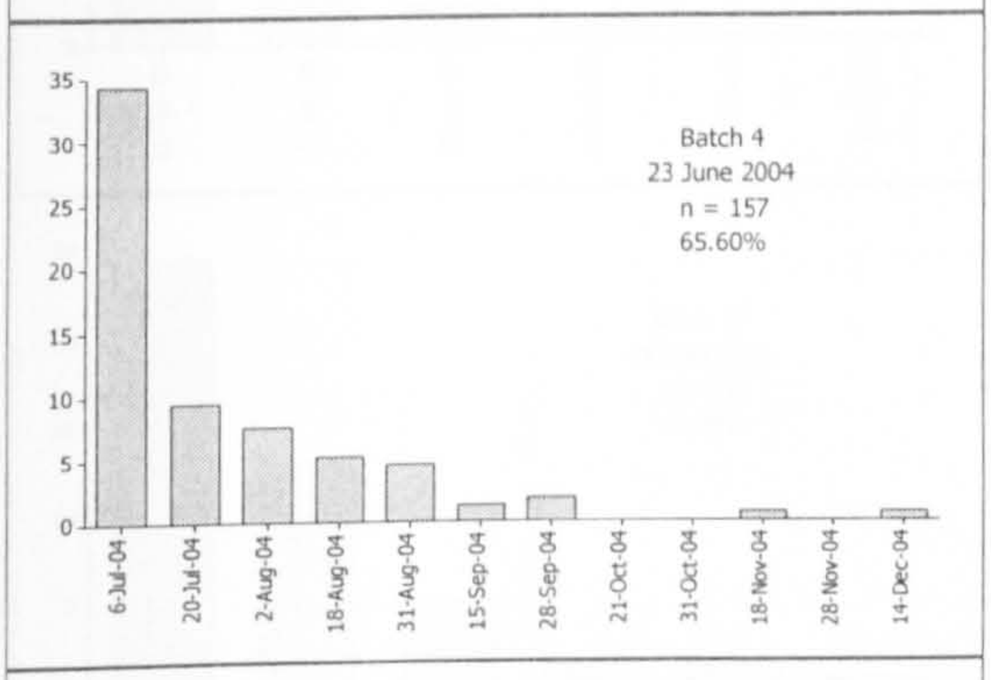


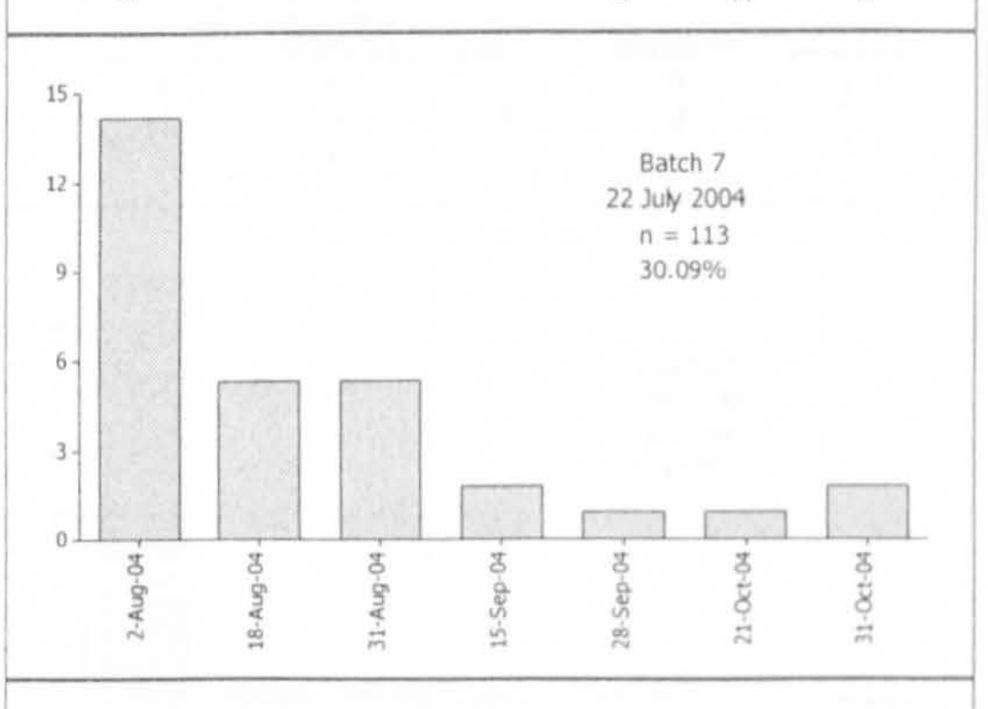
Figure 5.7. Percent (%) recovery of stocked *Scylla* spp. in monthly catches of crabs from the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines, June 2004-November 2005. Percent wild *Scylla* spp. (black bars) and percent recovered Scylla spp. (white bars) from the total monthly crab landings as follows: Jun 04 = 631, Jul 04 = 489, Aug 04 = 347, Sep 04 = 177, Oct 04 = 251, Nov 04 = 331, Dec 04 = 403, Jan 05 = 402, Feb 05 = 598, Mar 05 = 576, Apr 05 = 429, May 05 = 289, Jun 05 = 96, Jul 05 = 57, Aug 05 = 81, Sep 05 = 114, Oct 05 = 134, Nov 05 = 93.

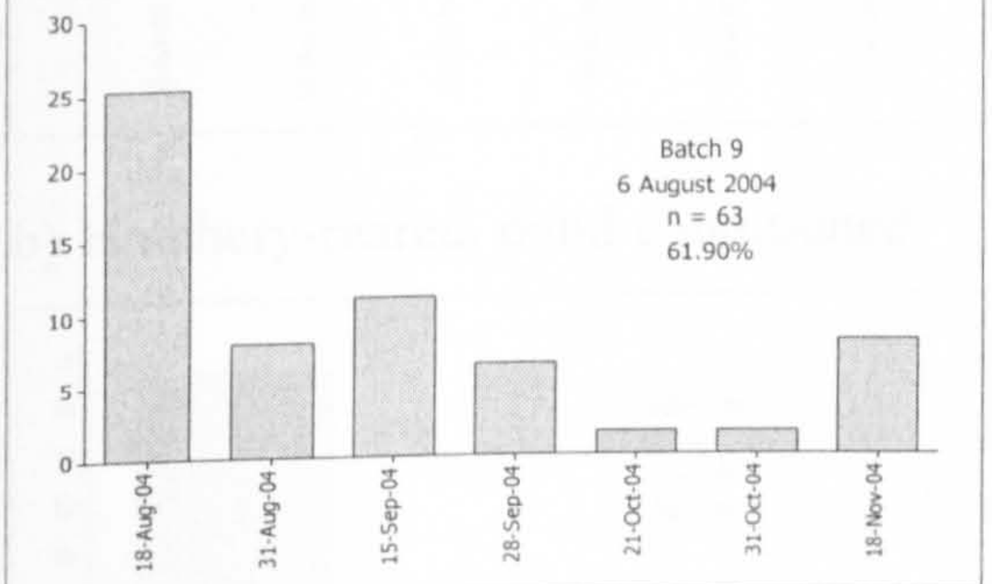
a) Wild-released

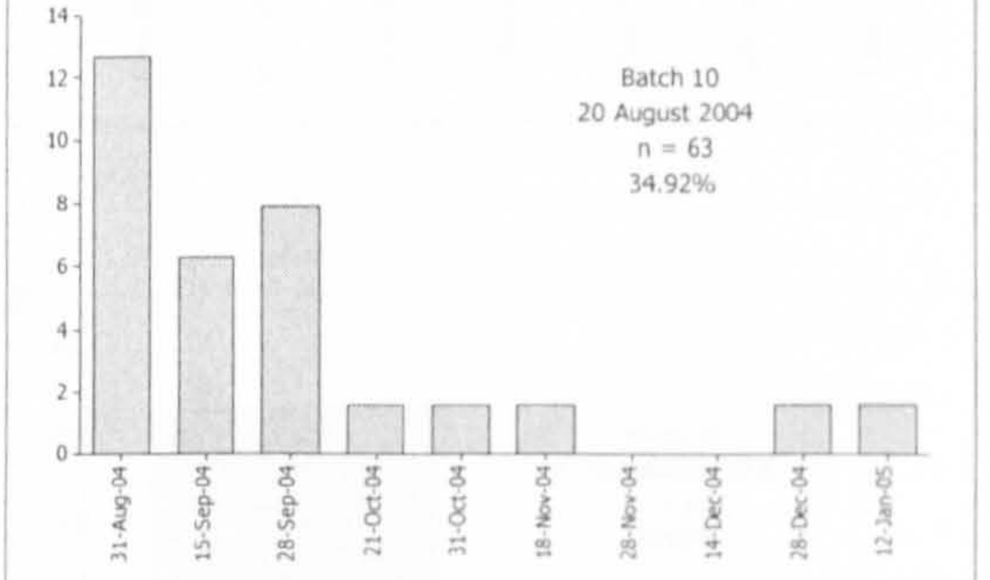


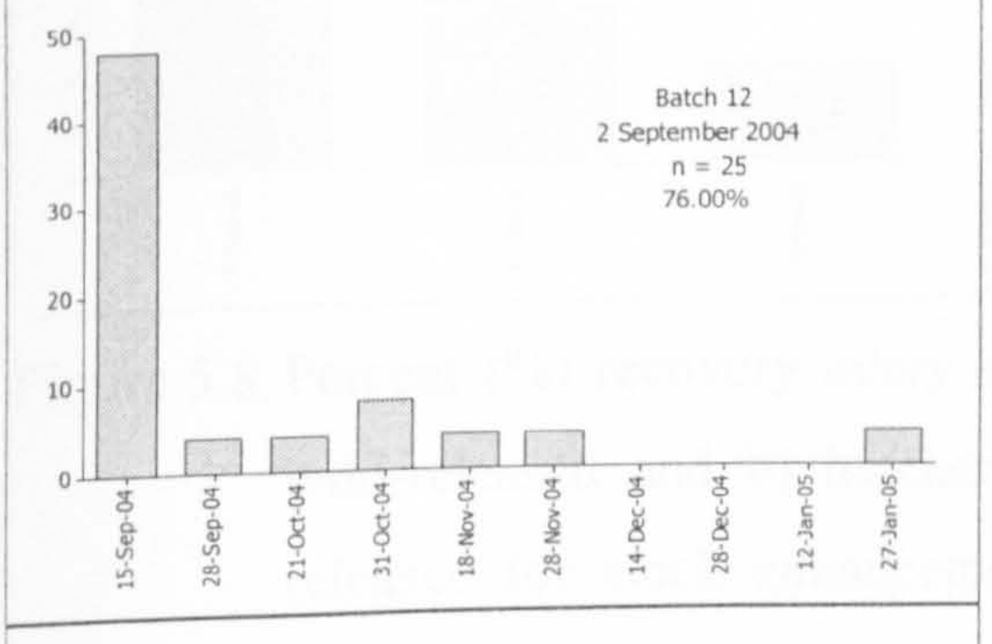


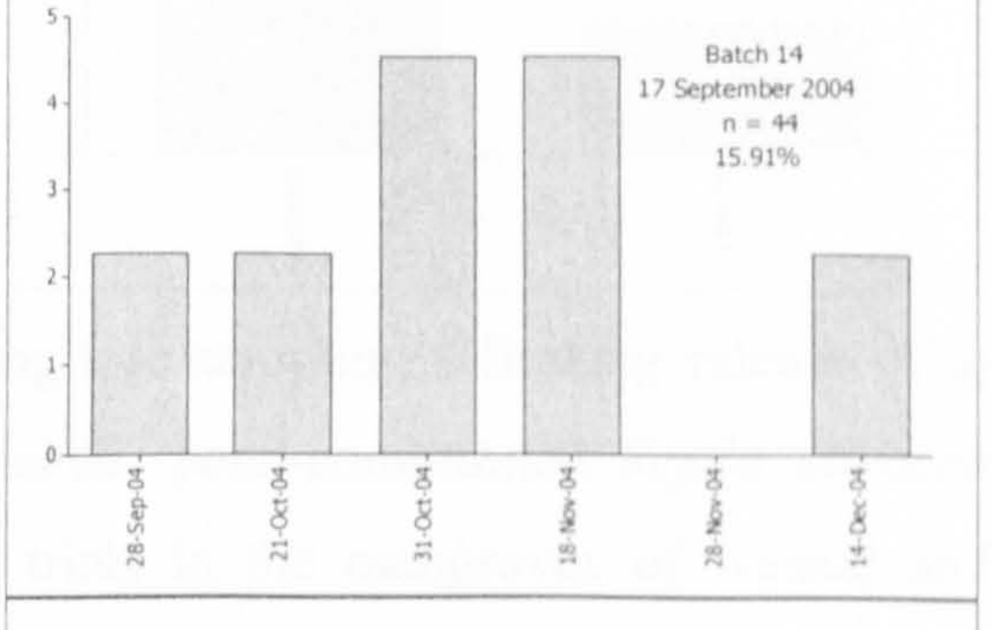


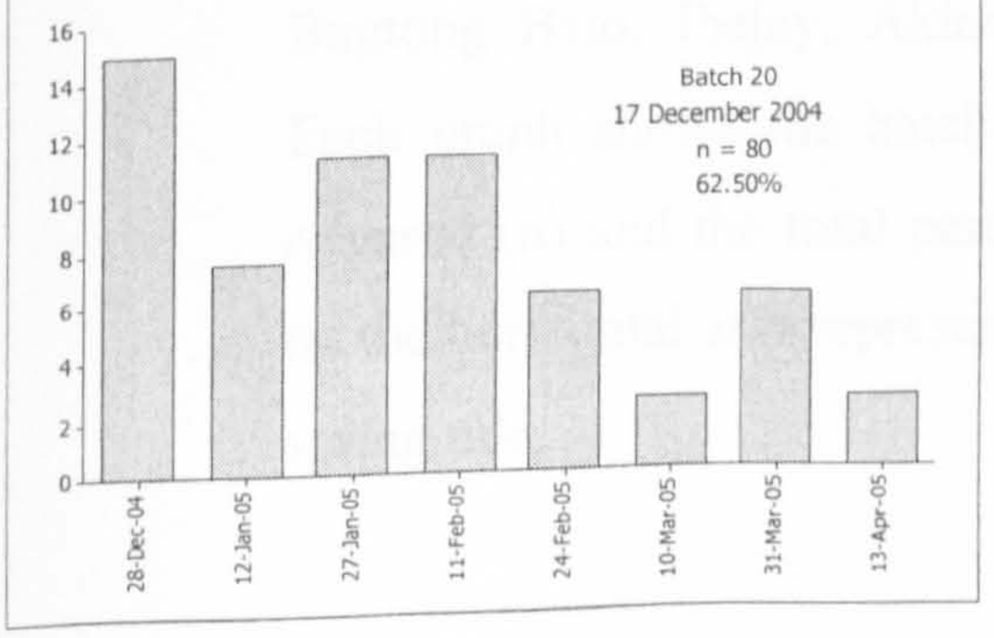


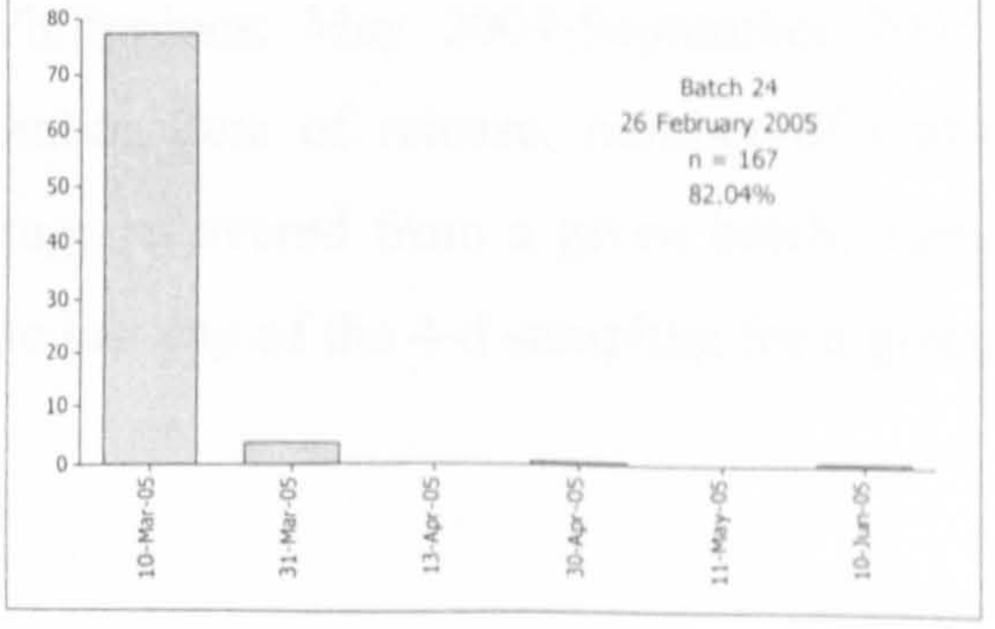


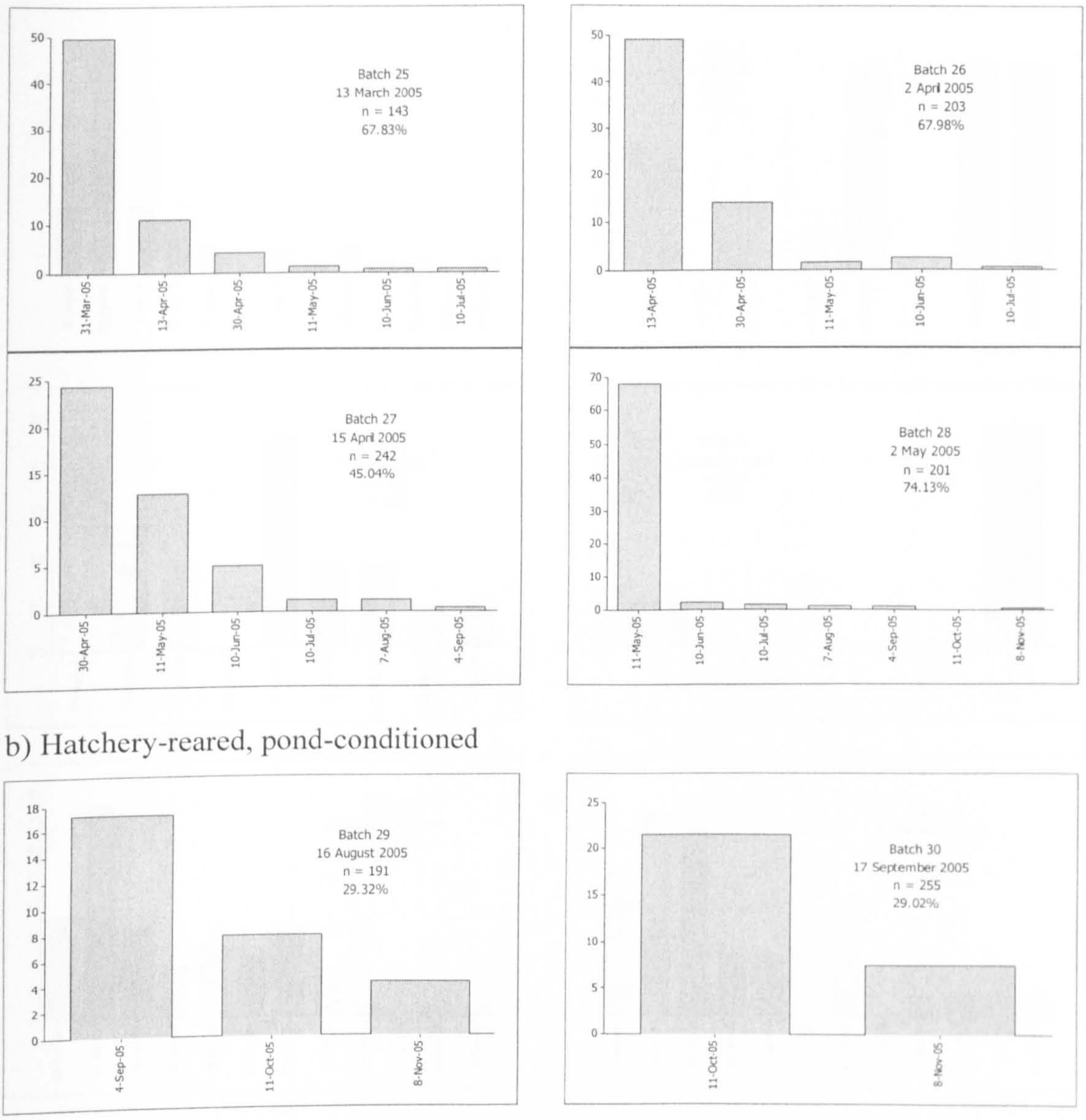








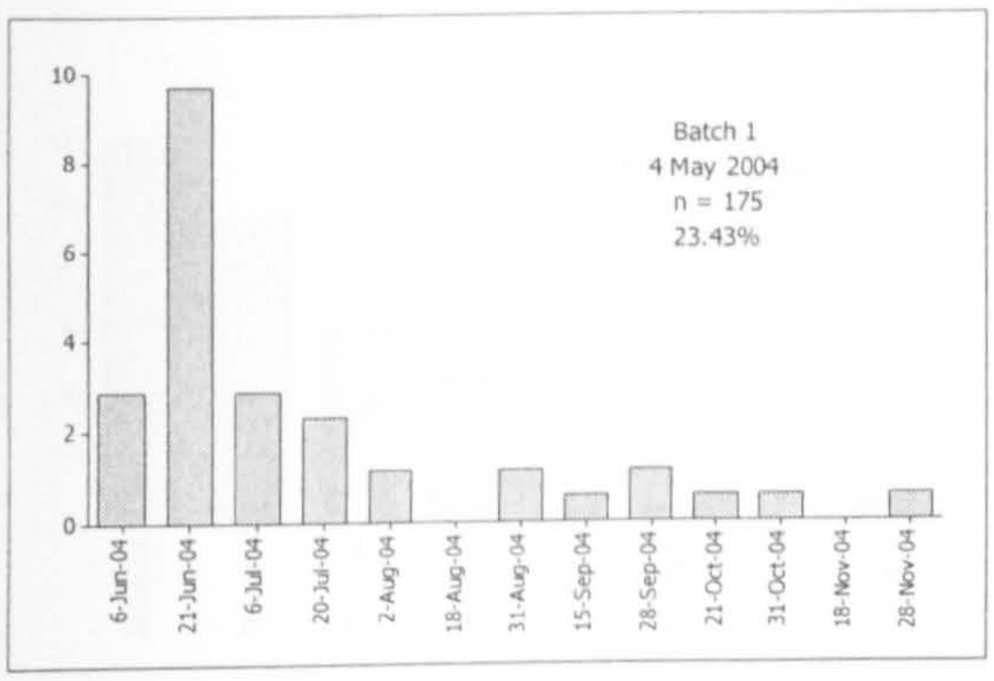


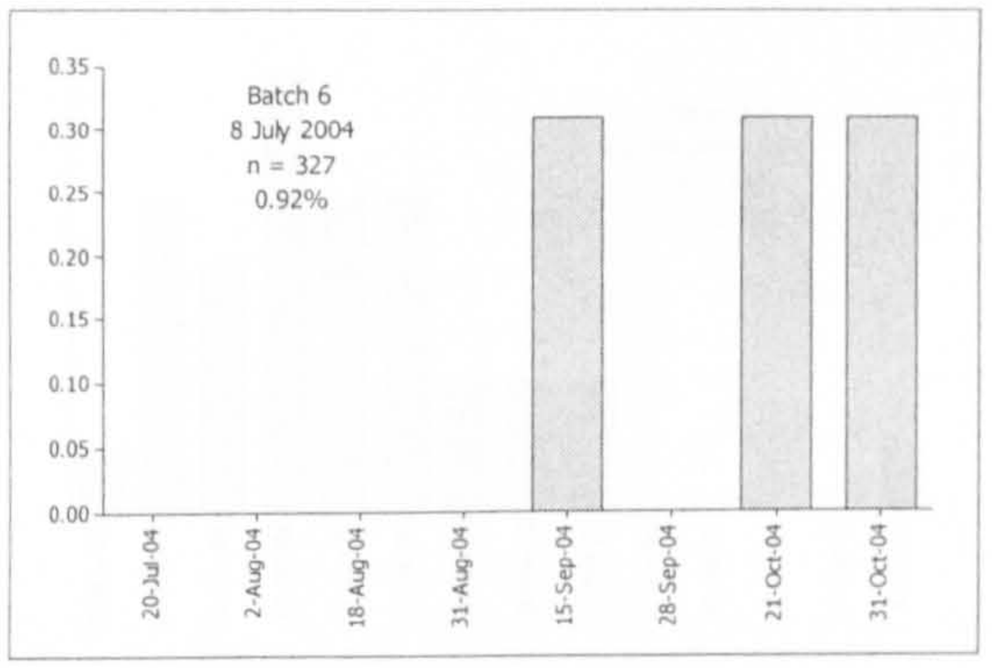


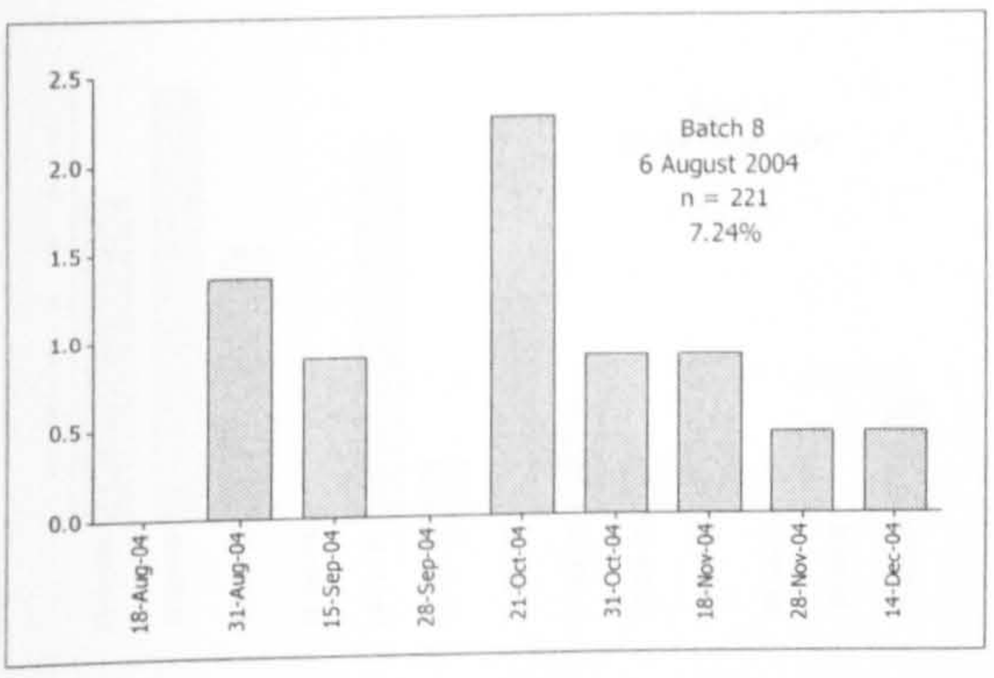
b) Hatchery-reared, pond-conditioned

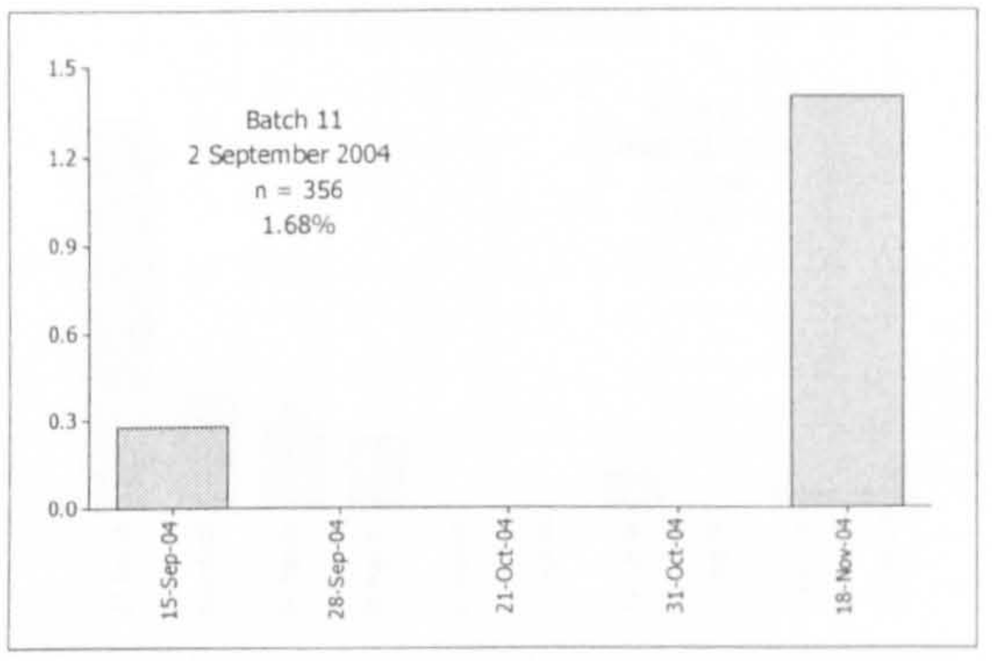
Figure 5.8. Percent (%) recovery every spring tide sampling following release of a) wild-released and b) hatchery-reared, pond-conditioned *Scylla olivacea* released for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; May 2004-September 2005. Each graph shows the batch number, date of release, number of crabs released (n) and the total percentage recovered from a given batch; dates on the horizontal axis represent the last day of the 4-d sampling for a given spring tide.

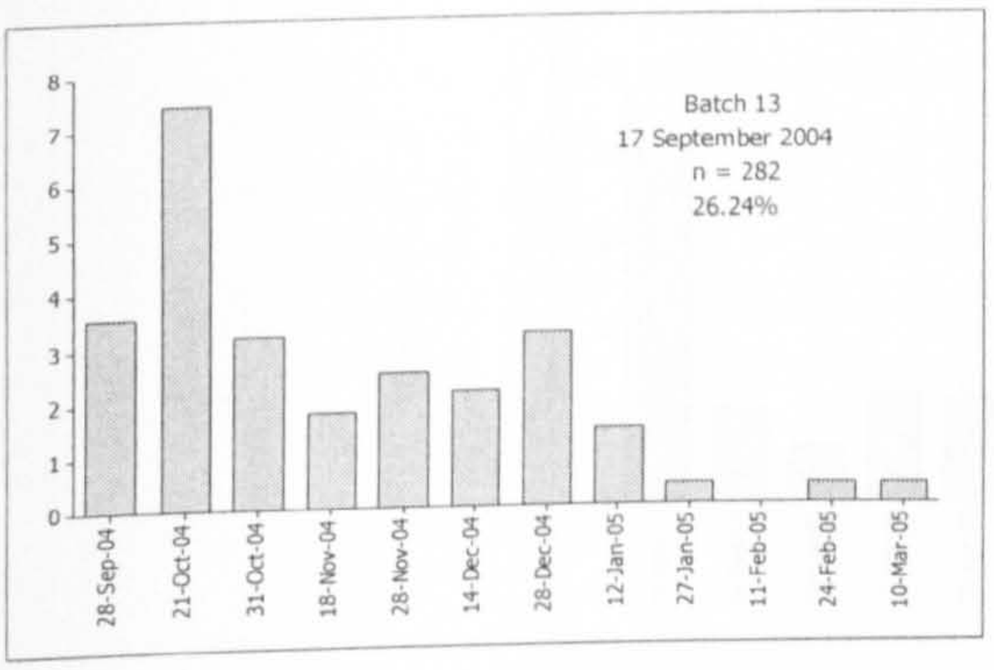
a) Hatchery-reared, unconditioned

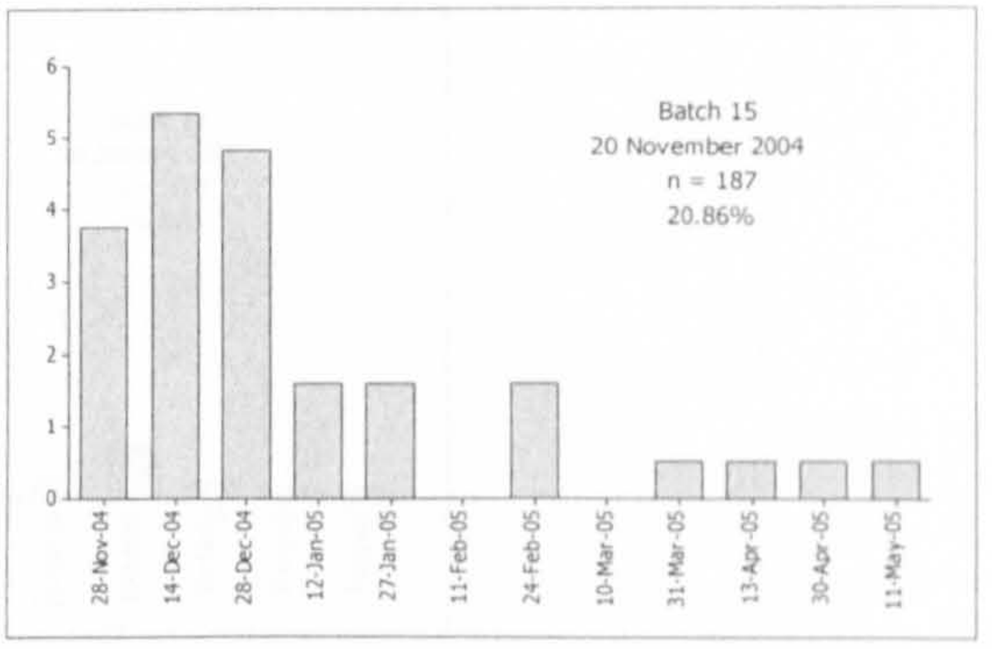


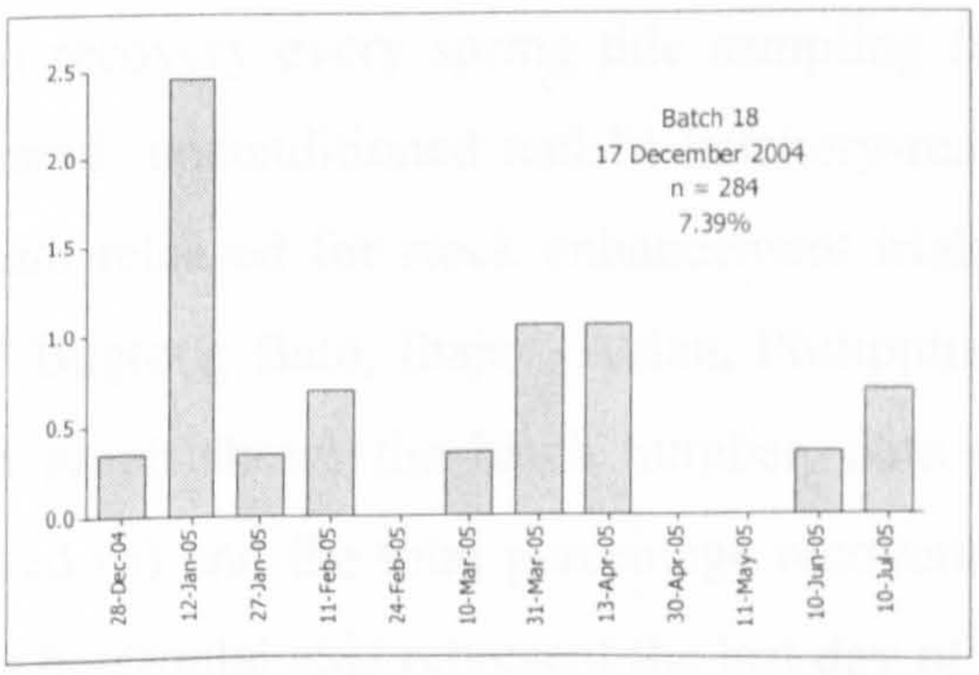




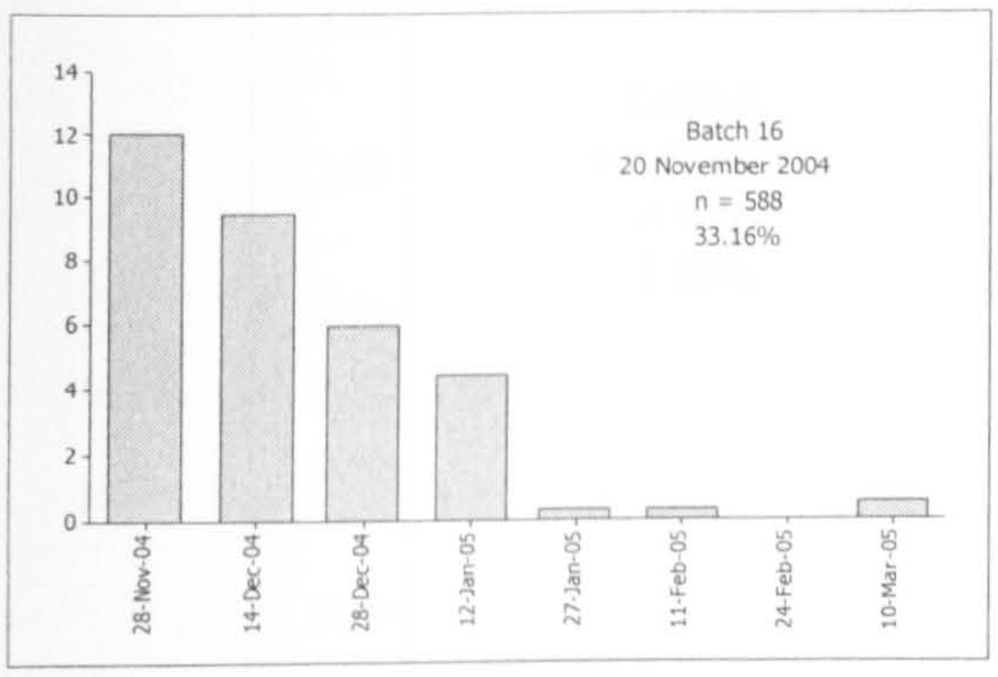


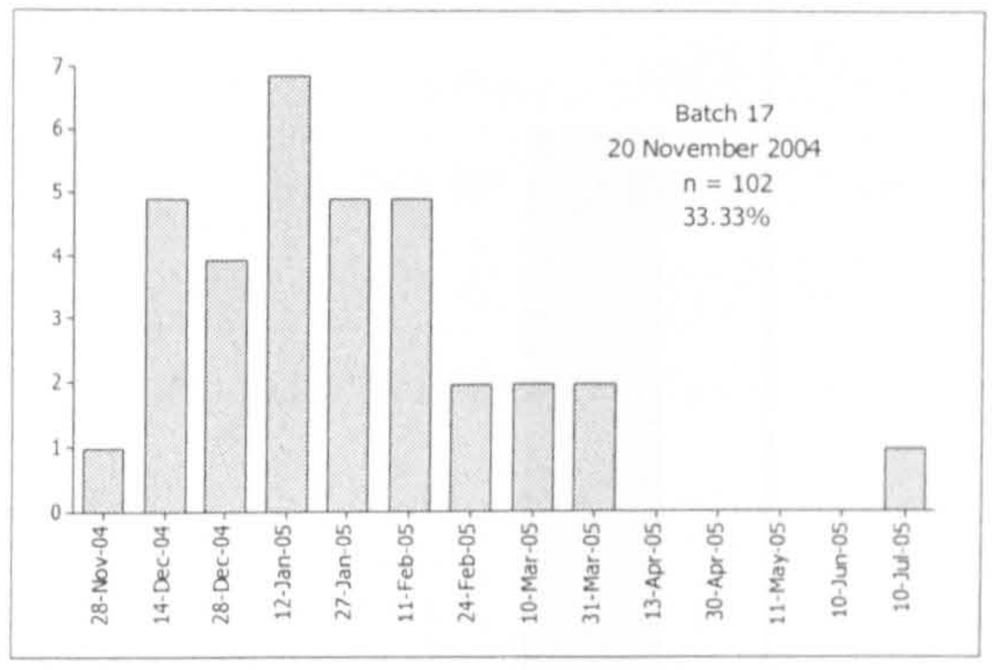


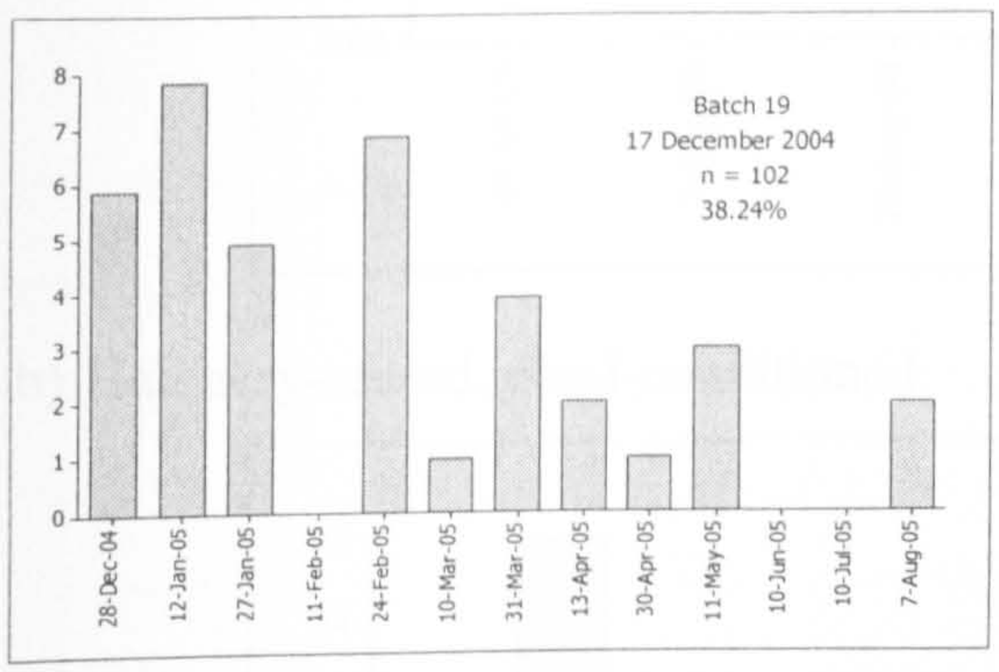


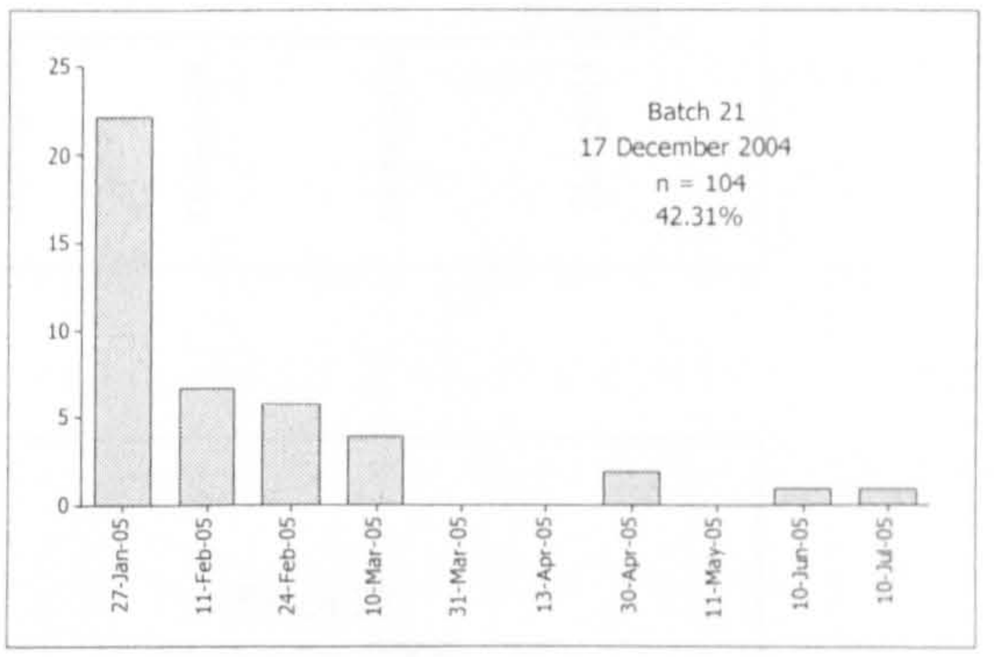


b) Hatchery-reared, pond-conditioned









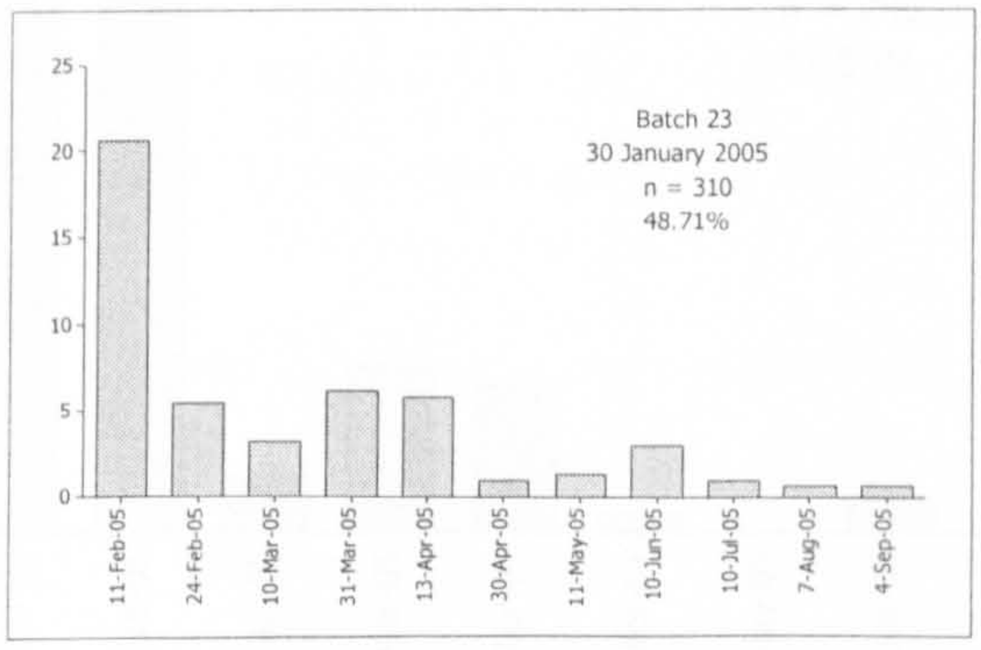
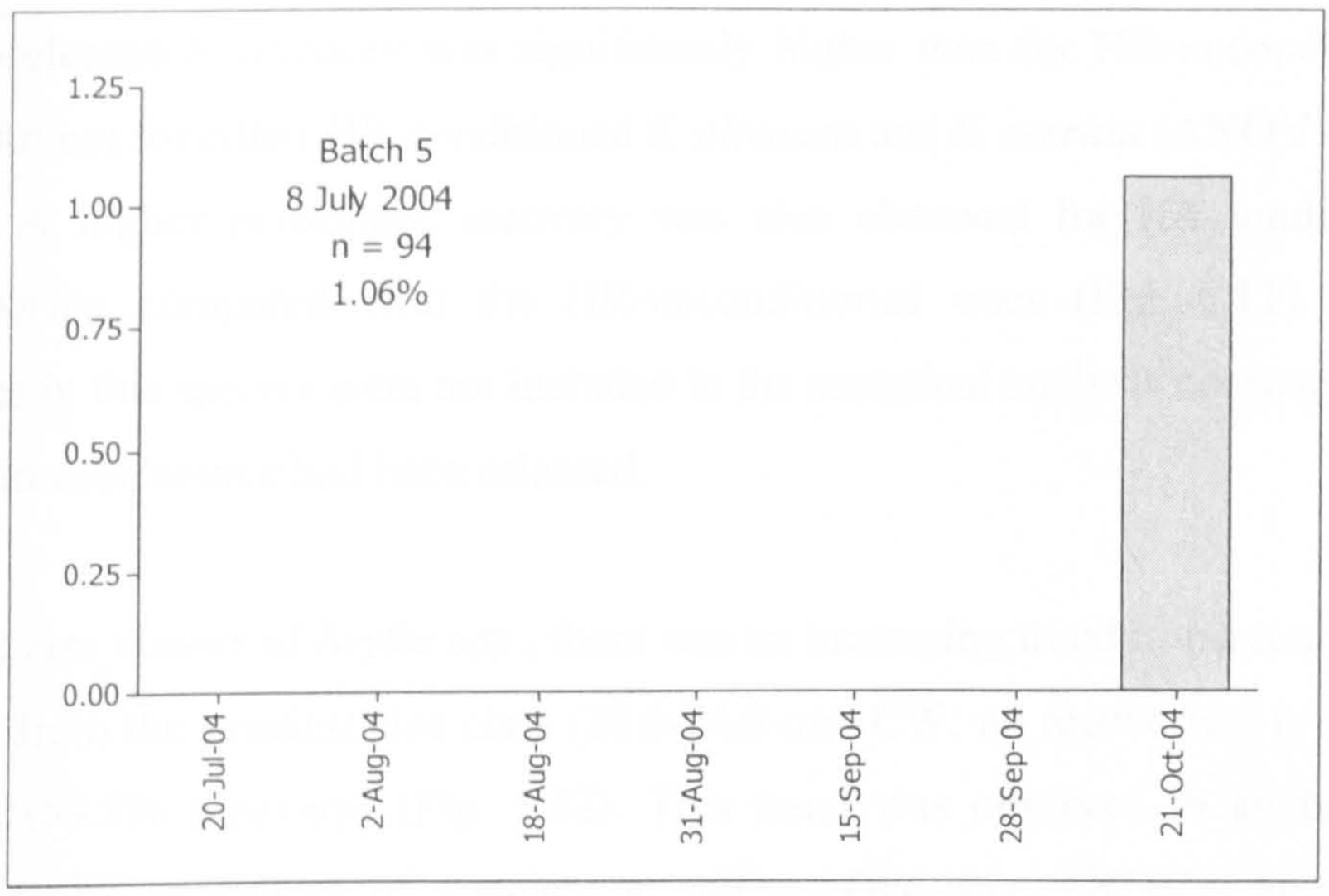


Figure 5.9. Percent (%) recovery every spring tide sampling following release of a) hatchery-reared, unconditioned and b) hatchery-reared, pond-conditioned *Scylla serrata* released for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; May 2004-January 2005. Each graph shows the batch number, date of release, number of crabs released (n) and the total percentage recovered from a given batch; dates on the horizontal axis represent the last day of the 4-d sampling for a given spring tide.

a) Hatchery-reared, unconditioned



b) Hatchery-reared, pond-conditioned

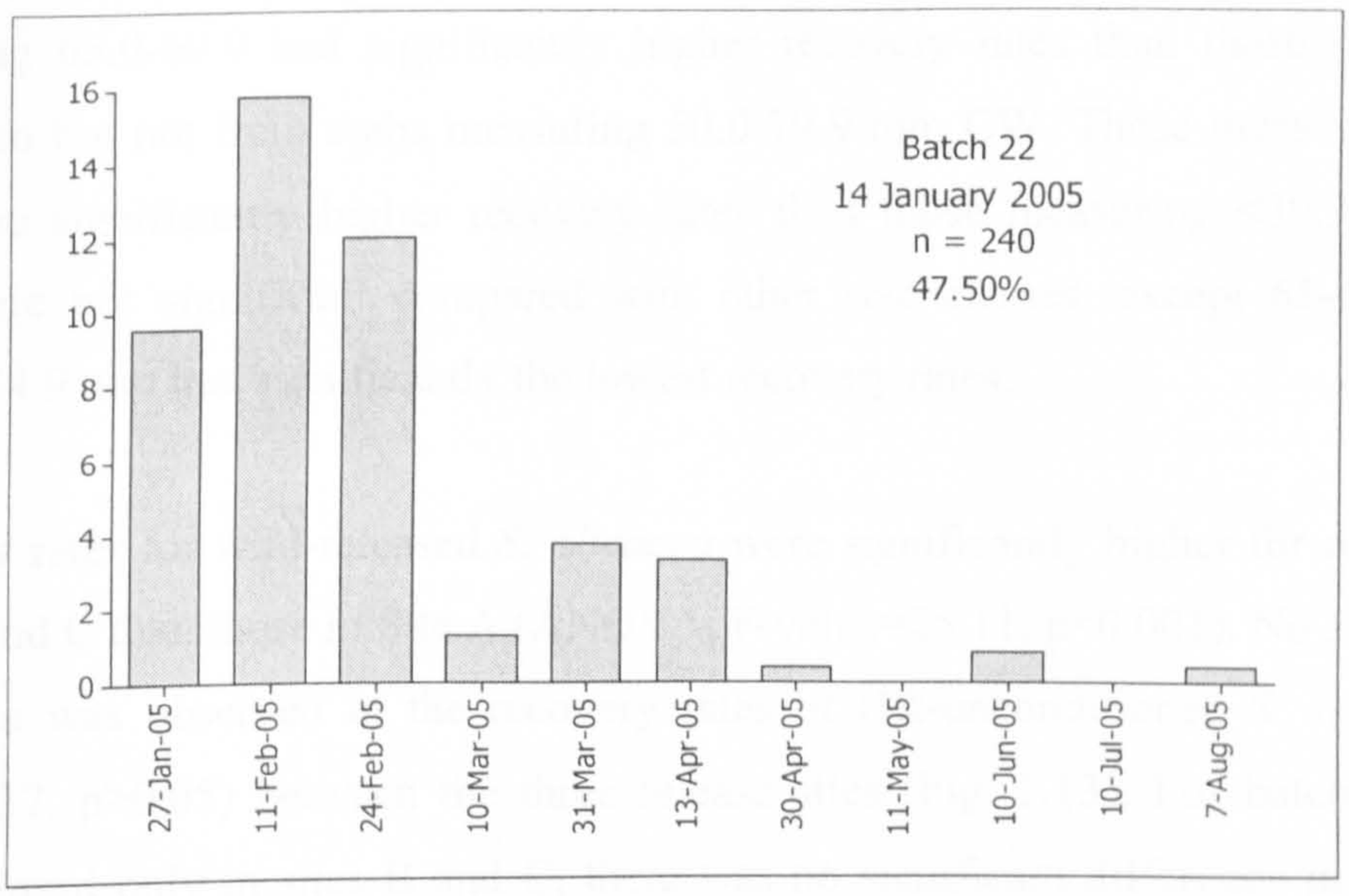


Figure 5.10. Percent (%) recovery every spring tide sampling following release of a) hatchery-reared, unconditioned and b) hatchery-reared, pond-conditioned *Scylla tranquebarica* released for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; July 2004 and January 2005. Each graph shows the batch number, date of release, number of crabs released (n) and the total percentage recovered from a given batch; dates on the horizontal axis represent the last day of the 4-d sampling for a given spring tide.

wild-released *S. olivacea* had the highest recovery rates (55.9 ± 6.3%) while HR-unconditioned *S. tranquebarica* had the lowest (1.1%) (Fig. 5.11). The recovery rate for wild-released *S. olivacea* was significantly higher than for HR-unconditioned *S. serrata* but not for either HR-conditioned *S. olivacea* and *S. serrata* (ANOVA, $F=6.36$, $p<0.01$). A higher percentage recovery was also observed for HR-conditioned *S. tranquebarica* compared with the HR-unconditioned ones (Fig. 5.11), although recoveries in this species were not included in the statistical analysis because only one batch from each source had been released.

Of the 12 size classes of *Scylla* spp., there was an increasing trend in percent recovery observed from the smallest size class (20.0-24.9 mm CW; no recoveries) to 65.0-69.9 mm CW (54.2% recovery) (Fig. 5.12). This trend was observed in all batches of released crabs regardless of species or source. One-way ANOVA showed that recovery rates significantly differ among size classes ($F=4.09$, $p<0.001$). Crabs measuring 65.0-69.9 had significantly higher recovery rates than those measuring <49.9 mm but not from crabs measuring 50.0-79.9 mm CW. Those measuring 45.0-49.9 have significantly higher recovery rates than those measuring <39.9 mm but these were not significant compared with other size classes, except 65-69.9 mm. Crabs <34.9 mm had significantly the lowest recovery rates.

Recovery rates for wild-released *S. olivacea* were significantly higher for releases in Sites B and C than those in Site A (ANOVA, F-value=25.11, $p<0.001$). No significant difference was observed in the recovery rates of HR-unconditioned *S. serrata* (F-value=5.37, $p>0.05$) between the three release sites (Fig. 5.13). For batches which were released only in sites B and C, there was no significant difference in recovery rates between these two release sites. Release of crabs is Site A was stopped after September and transferred to Site C after very low initial recovery rates were observed for batches of crabs released in Site A. The presence of a large aquaculture pond near the release site (see Fig. 5.1) may have affected the survival of the crabs due to constant draining of pond water into the area which may have contained pollutants detrimental to crabs. Incidence of fish kills in the vicinity of these ponds occurred twice within the four-yr duration of the study.

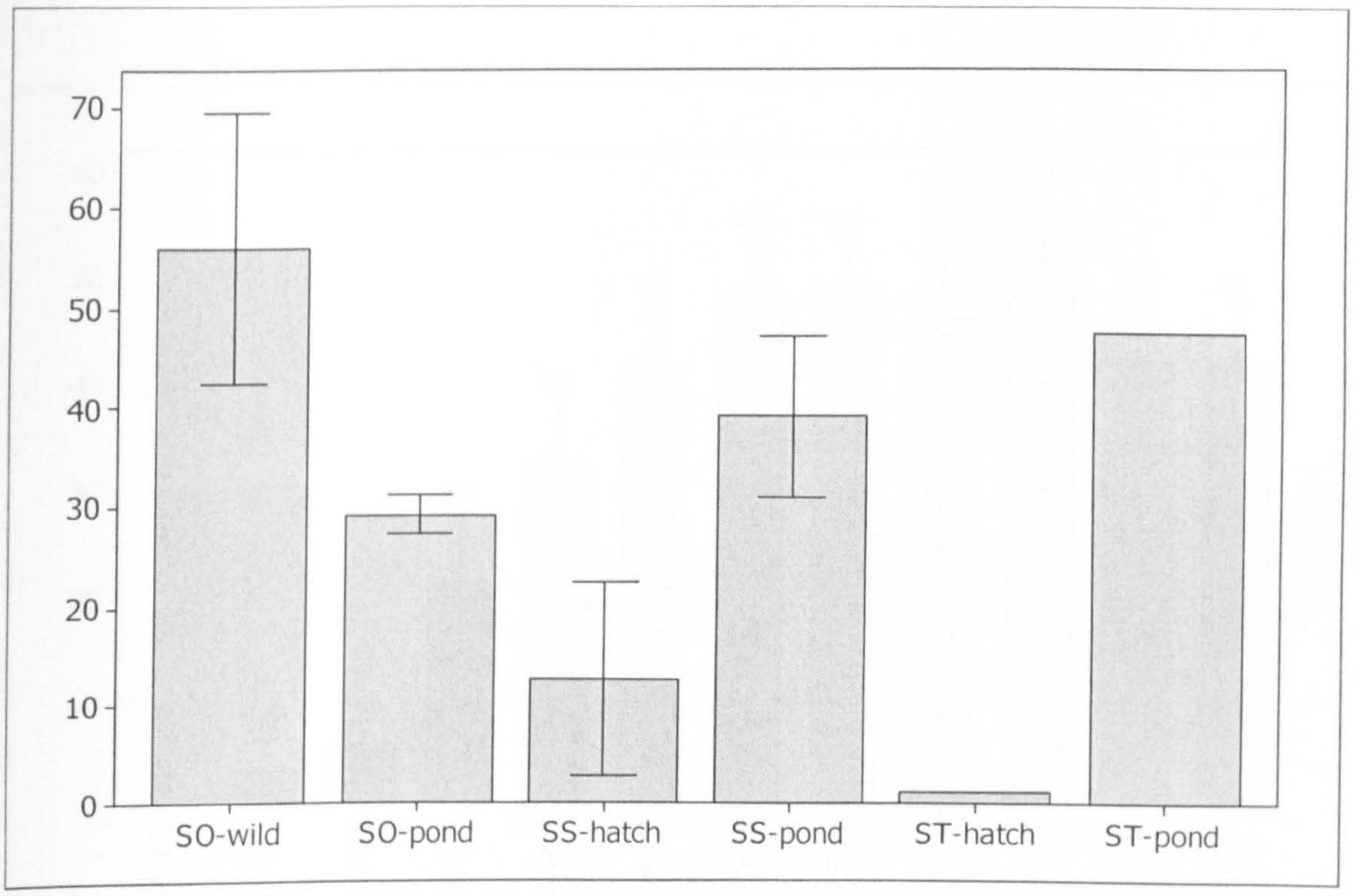


Figure 5.11. Percent (%) mean recovery of *Scylla olivacea*: wild-released (SO-wild) and hatchery-reared, pond-conditioned (SO-pond), *Scylla serrata*: both hatchery-reared, unconditioned (SS-hatch) and pond-conditioned (SS-pond), and *Scylla tranquebarica* (only one sample each): both hatchery-reared, unconditioned (ST-hatch) and pond-conditioned (ST-pond) released for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; May 2004-September 2005. Error bars represent 95% confidence interval from the mean. Number of crabs released (N) are shown in Table 5.1.

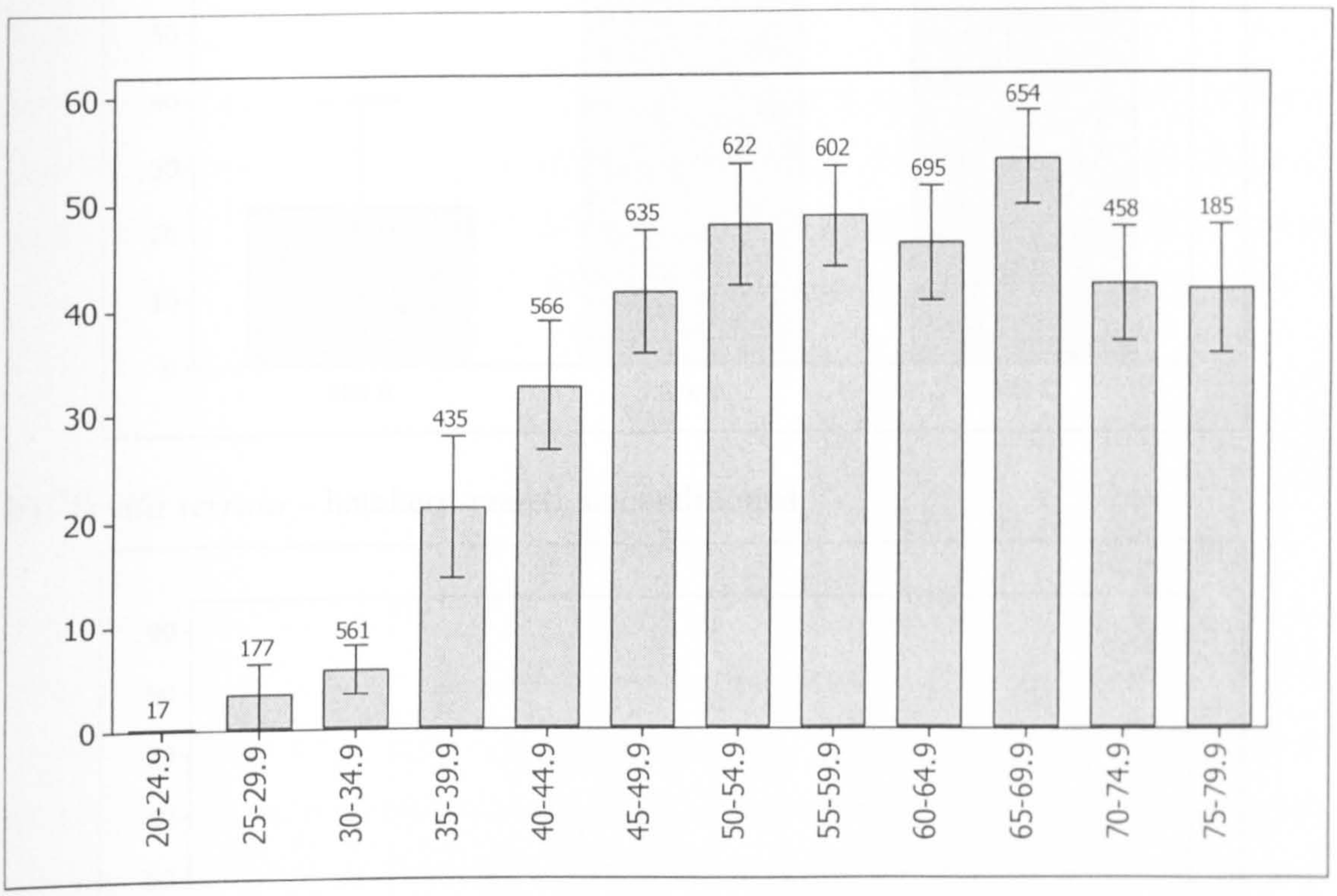
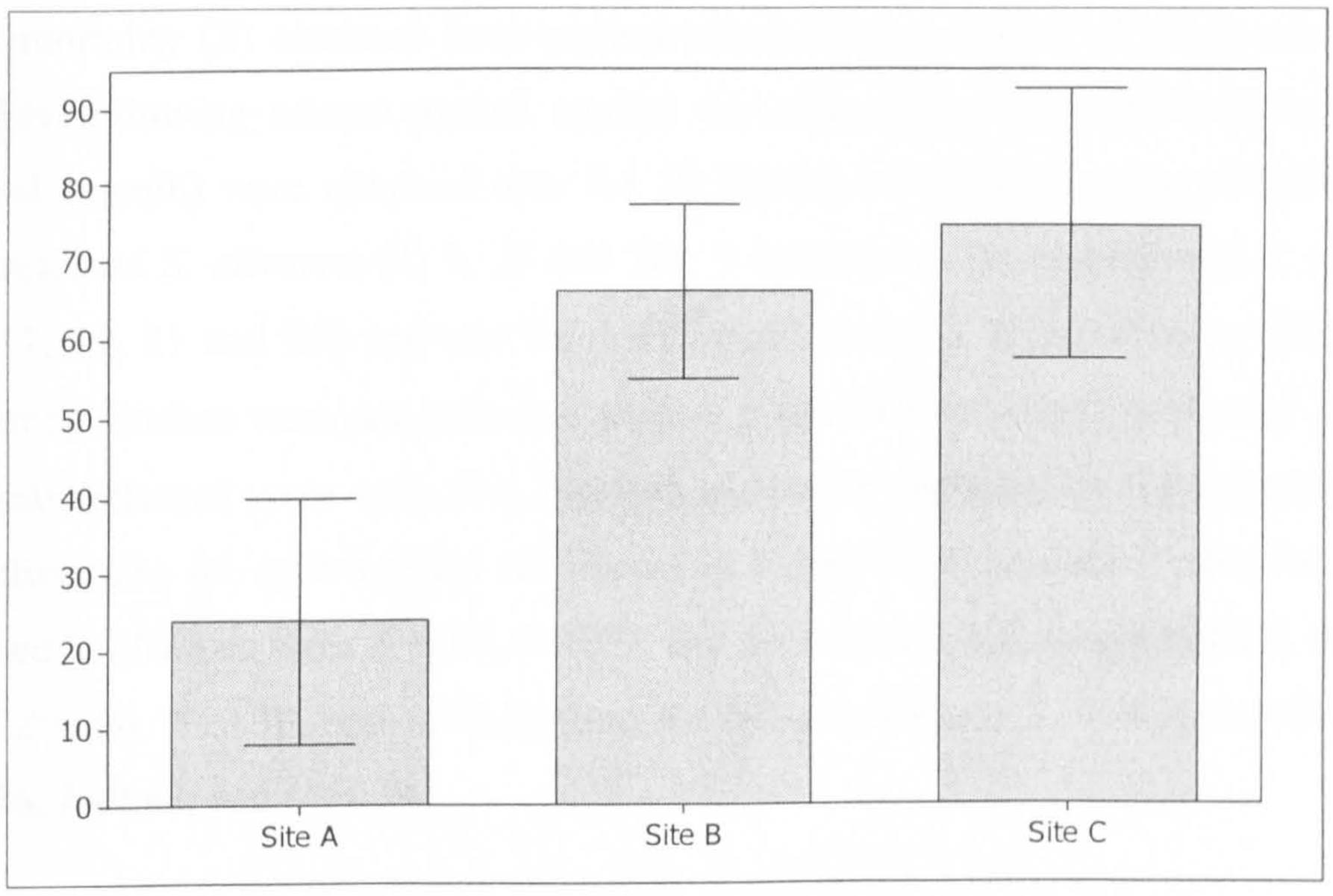


Figure 5.12. Percent (%) mean recovery of *Scylla* spp. from different size-classes (in mm CW) released for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; June 2004-November 2005. Error bars represent 95% confidence interval from the mean. Number of crabs released per size class are shown on top of each error bar.

a) *Scylla olivacea* – wild-released



b) *Scylla serrata* – hatchery-reared, unconditioned

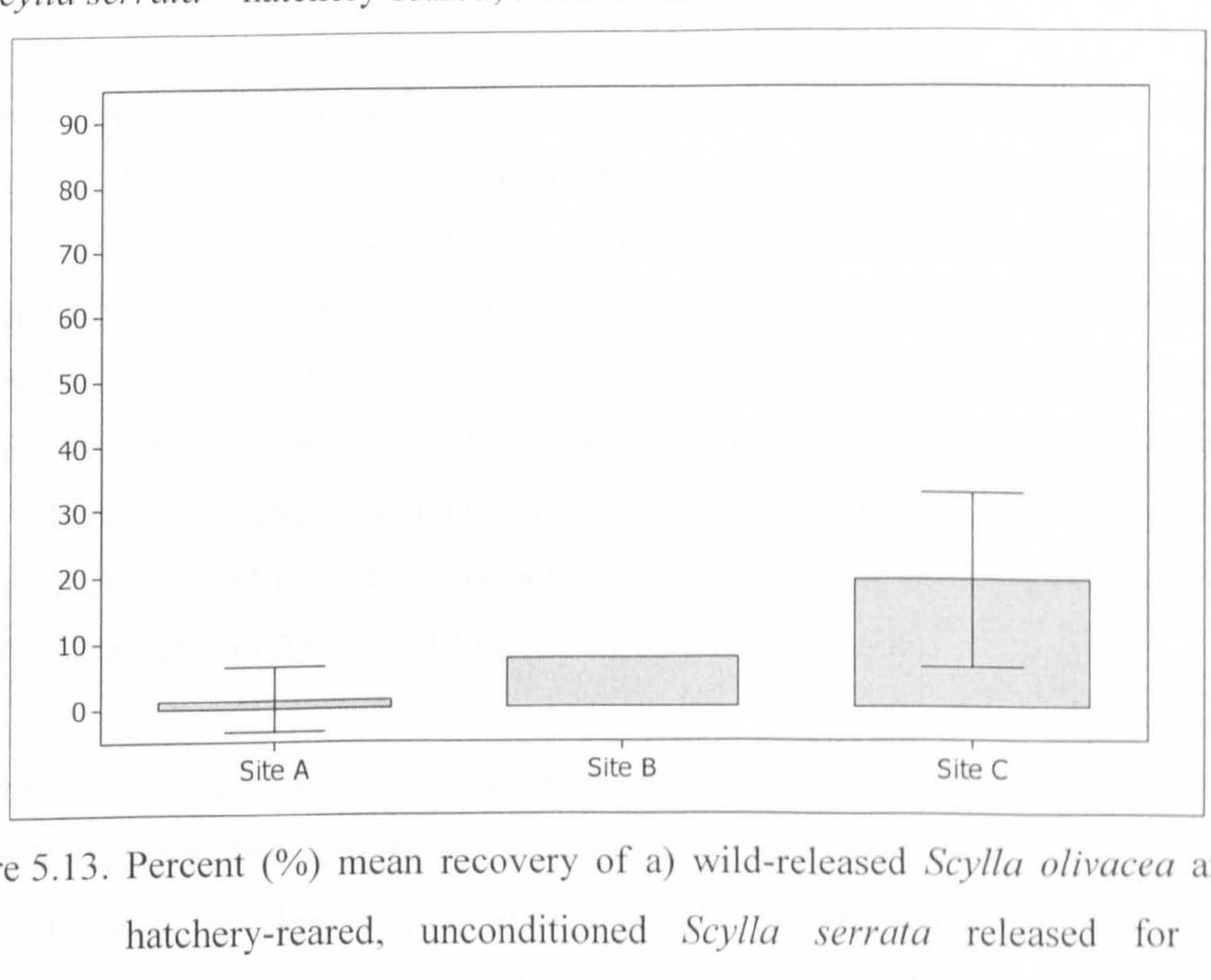


Figure 5.13. Percent (%) mean recovery of a) wild-released *Scylla olivacea* and b) hatchery-reared, unconditioned *Scylla serrata* released for stock enhancement trials in different sites in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; June 2004-November 2005. Error bars represent 95% confidence interval from the mean. Number of crabs released per site (N) are shown in Table 5.1.

Mortality

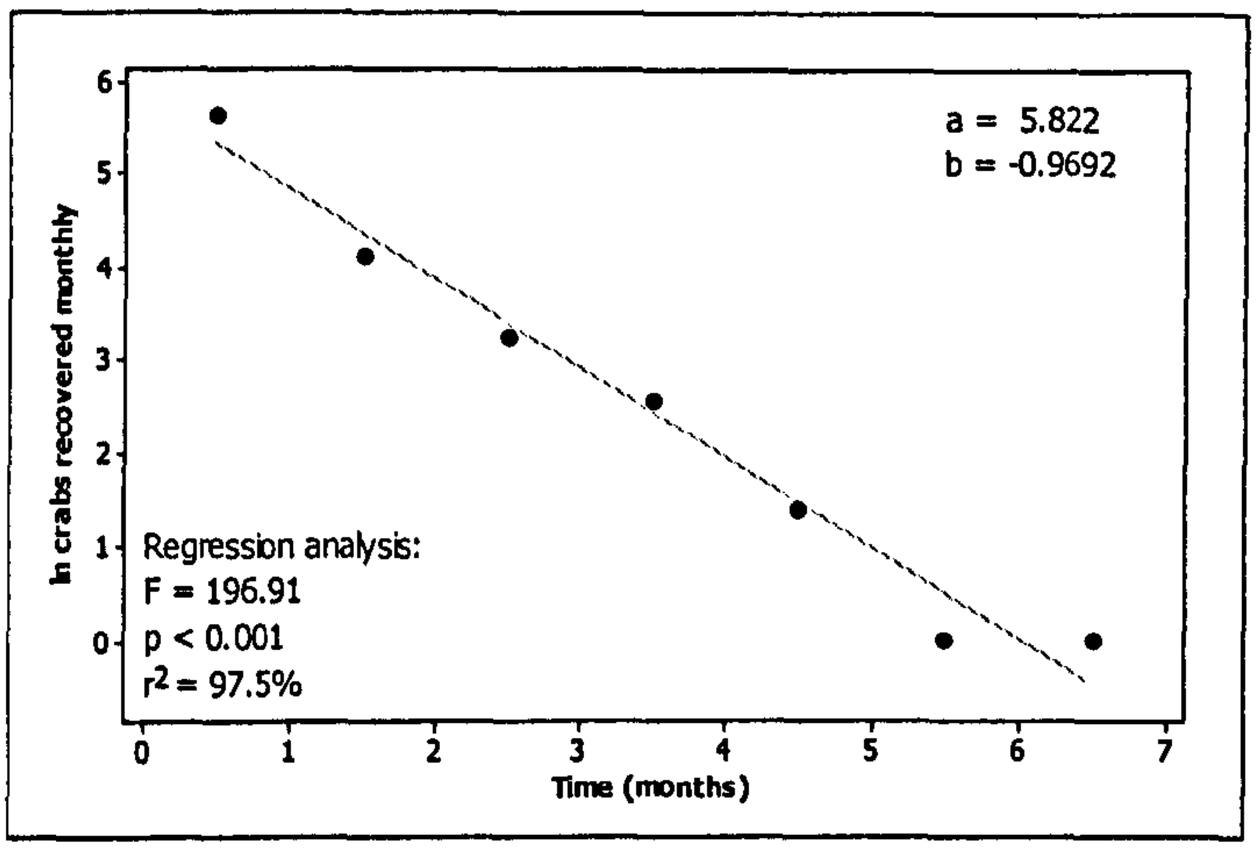
Total mortality (Z) obtained from pooled values of the number of crabs recovered monthly following release plotted against the midpoint of the corresponding time interval (month) were obtained only for 10 batches of released crabs; 4 batches of wild-released *S. olivacea* (4, 7, 27 and 28), 5 batches of HR-conditioned *S. serrata* (16, 17, 19, 21 and 23) and one batch HR-conditioned *S. tranquebarica* (22). The remaining batches were not included because either recovery rates were very low or the crabs released were very few. Plots of regression analyses for the estimation of mortality rates for each species are shown in Figure 5.14. Mortality rates for wild-released *S. olivacea* were Z=0.97, F=0.74, and M=0.23; for HR-conditioned *S. serrata* were Z=0.66, F=0.30, and M=0.36; and for HR-conditioned *S. tranquebarica* were Z=0.76, F=0.42, and M=0.34.

Movement

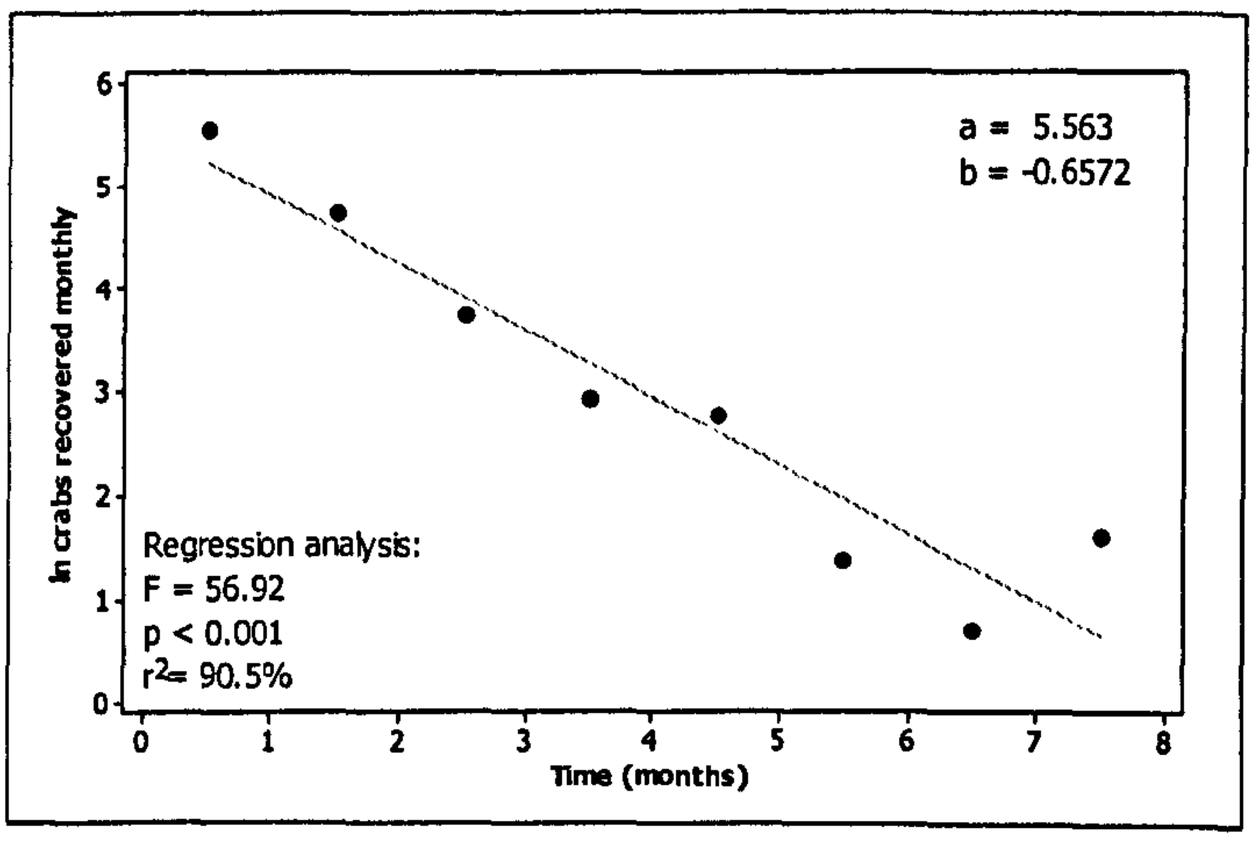
Movement of crabs from one area to the other is illustrated in Figures 5.1 (release sites and collection areas). In general, the results of this study showed that most of the crabs regardless of species and source, tended to stay close to where they were released or just moved around the vicinity of the release site (Fig. 5.15). Of the crabs released in Site A, at least 80% of the recovered crabs both for wild-released *S. olivacea* and HR-unconditioned *S. serrata* remained in the same creek where they were released (collection Area 1). The greatest distance moved was to Area 5. The whole creek covered by Area 1 measured 600 m while the distance between release Site A and collection Area 5 is 1 km.

Of the crabs released in Site B, most of the wild-released *S. olivacea*, HR-unconditioned and HR-conditioned *S. serrata* were recovered along the 600-m stretch creek in Area 3. However, most of the HR-conditioned *S. olivacea* and *S. tranquebarica* moved downstream along the main creek in Area 2 where more than 50% of them were recovered. From the release Site B, a maximum of 1.4 km have been moved by crabs caught downstream of Area 2, just before Area 5 where the three creeks meet.

a) *Scylla olivacea*



b) *Scylla serrata*



c) *Scylla tranquebarica*

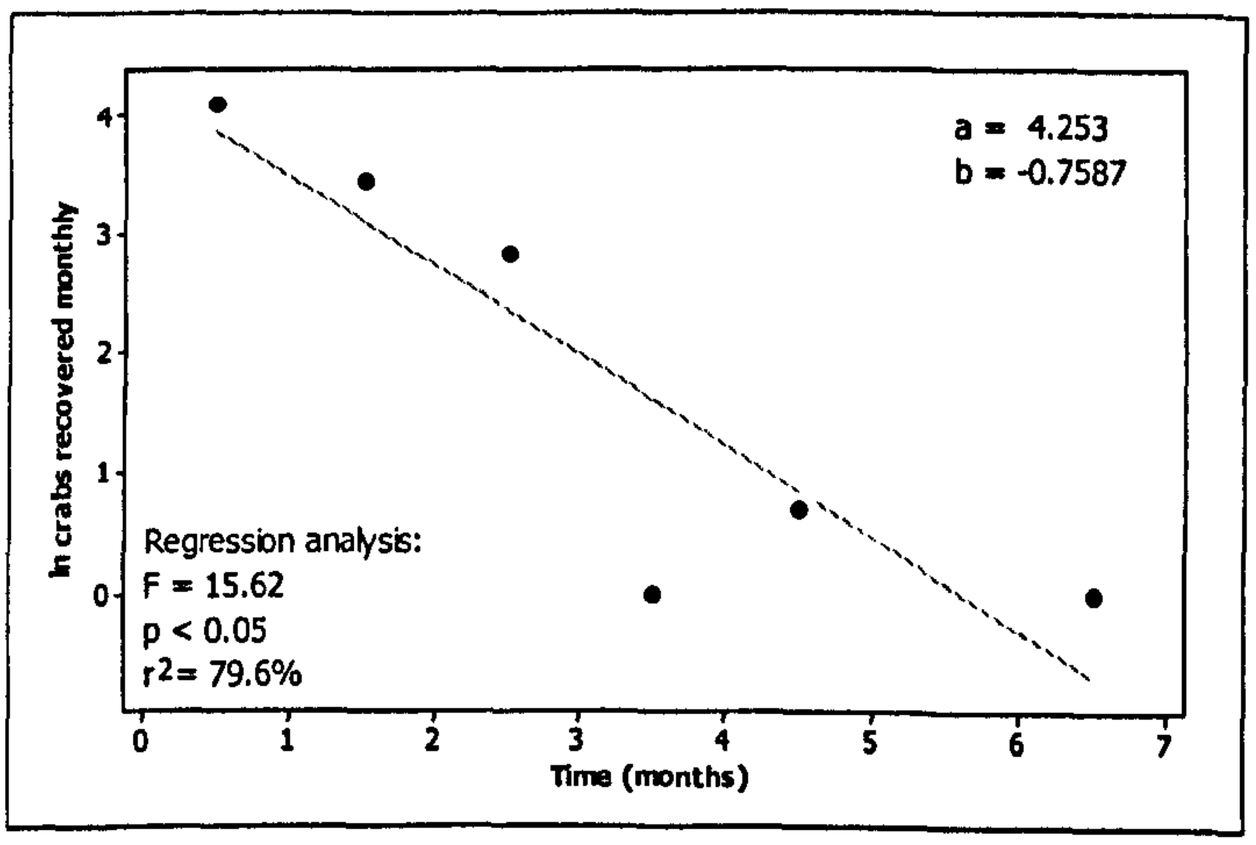


Figure 5.14. Regression analysis plots showing values of the intercepts (a) and slopes (b) used to estimate total (Z) and fishing (F) mortalities for a) *Scylla olivacea*, b) *Scylla serrata*, and c) *Scylla tranquebarica* released for stock enhancement trials in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; May 2004-September 2005.

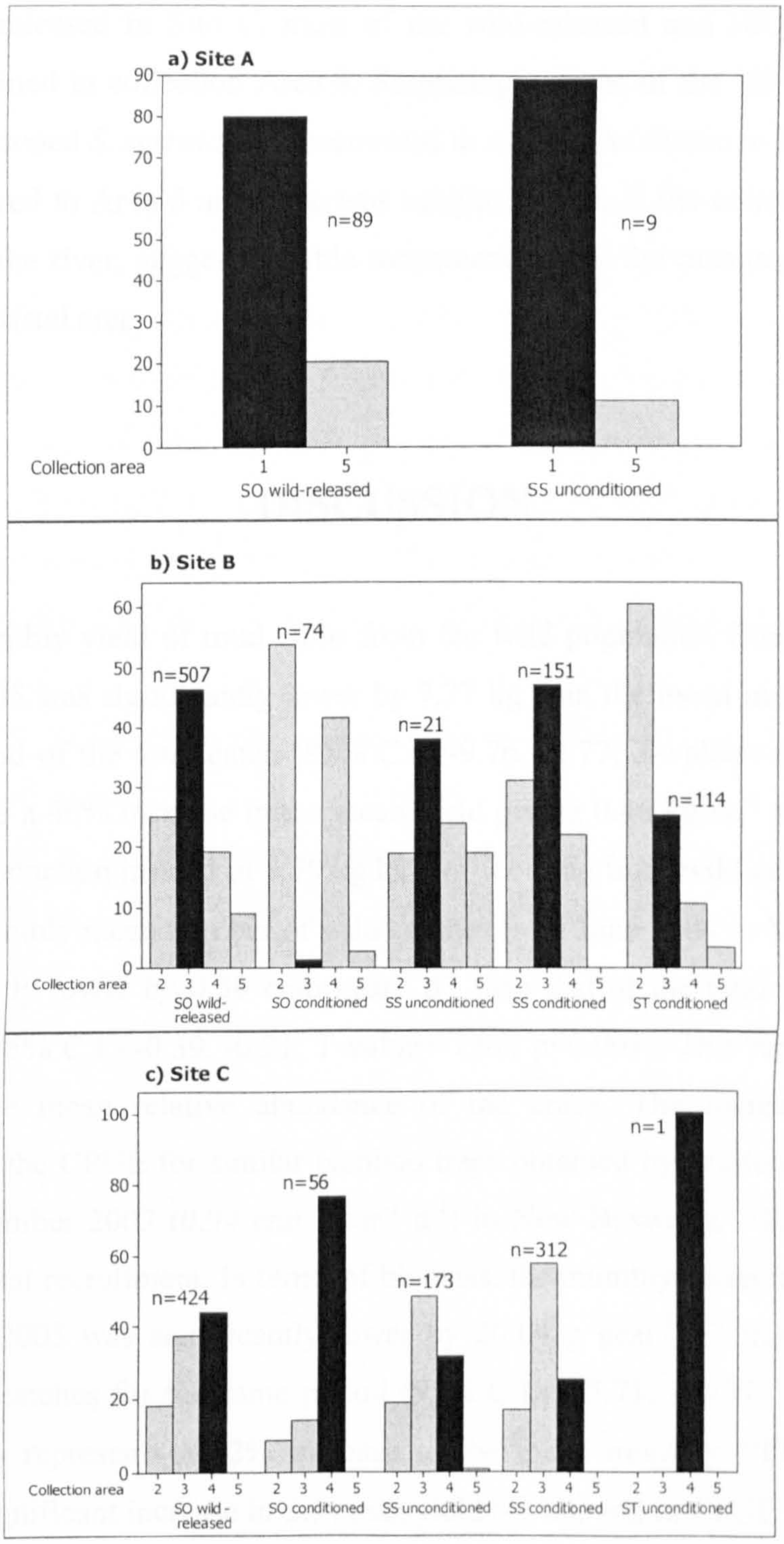


Figure 5.15. Percentage (%) of recovery of *Scylla* spp. released in a) Site A, b) Site B, and c) Site C and recovered from different collection areas in the mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines; June 2004-November 2005. Species (SO = *S. olivacea*, SS = *S. serrata*, ST = *S. tranquebarica*) and sources are shown below each cluster of collection areas; black bars represent collection area nearest the release site; n the number of crabs recovered.

Of the crabs released in Site C, most of the wild-released and HR-conditioned *S. olivacea* remained in collection Area 4. Surprisingly, most of the HR-unconditioned and HR-conditioned *S. serrata* were recovered in Area 3. As shown in Figure 16, very few crabs moved to Area 5 and none was caught in Area 6, the collection area near the mouth of the river, suggesting little movement out of the mangrove system into the adjacent coastal area.

DISCUSSION

The mean monthly yield of mud crabs from the wild population from June 2004 to November 2005 was significantly lower by 7.27 kg than the mean monthly yield for the same period of the total catch (95% C.I.=-9.76, -4.77; T-value=-6.15; $p<0.001$). This represents a 46% increase in the mean yield giving 0.46 kg ha^{-1} $month^{-1}$ or 5.54 kg ha^{-1} yr^{-1} production instead of 3.79 kg ha^{-1} yr^{-1} coming from wild catches only. For CPUE, the monthly mean number of wild catches from June 2004 to November 2005 was significantly lower by 0.30 crab $gear^{-1}$ d^{-1} than that of the total catches for the same period (95% C.I.=-0.39, -0.21; T-value=-7.06; $p<0.001$). This represents a 51% increase in the mean relative abundance of the crabs. The increased CPUE is comparable to the CPUE for similar bamboo traps obtained by Walton et al. (2006a) for April-December 2003 (0.94 crab $gear^{-1}$ d^{-1}) in New Buswang, Aklan mangroves with high natural recruitment. In terms of biomass, the monthly mean from June 2004 to November 2005 was significantly lower by 20.04 g $gear^{-1}$ d^{-1} than the monthly mean of total catches for the same period (95% C.I.=-25.71, -14.37; T-value=-7.46, $p<0.001$). This represents a 42% increase in the mean monthly CPUE (biomass). There was a significant increase in the mean yield (46%), and in CPUE number (51%) and biomass (42%) resulting from the crabs released into the mangroves.

Similarly, large contributions of hatchery-reared animals to stocks have been observed in some other species. In 1995, 40% of the young lobster *Homarus gammarus* population in Norway were estimated to originate from the 155,000 juveniles released since 1988 (Svasand and van der Meeren 1995). Agnalt et al. (1999) further reported that experimental releases of cultured juveniles have

substantially augmented depleted lobster population in Norway, comprising up to 40% of the total landings in 1997. From pilot releases of cultured mullet *Mugil cephalus* in 1990-1993 in Hawaii, cultured fish comprised 13% of the commercial mullet catch in autumn of 1994 (Leber and Arce 1996). Moreover, Leber and Lee (1997) reported that those releases made significant contributions to the commercial striped mullet fishery in Kaneohe Bay from 3% following 1990 releases, 10% after 1991 releases and 50% in the present study. In the early 1990s, percent contribution of enhanced production of Pacific salmon *Oncorhynchus* spp. in Canada increased from 10 to 29% for Chinook *O. tshawytscha*, 9 to 13% for coho *O. kisutch*, 3 to 18% for pink *O. gorbuscha* and remained high at 40% for chum *O. keta* (Beamish and Noakes 2004). In Japan, the 30-yr stock enhancement efforts on the swimming crab *Portunus trituberculatus* are responsible for the 9% contribution to crab fisheries in Okayama prefecture, 18% in Hamana Lake, and 59% in Osaka Bay (Secor et al. 2002). Furthermore, in Kagoshima Bay, commercial landings from 1989 to 1991 were mainly composed of released red sea bream *Pagrus major*, from 30% in the central part of the bay to 64-83% in the inner part of the bay (Fushimi 2001).

In the present work, although a 46% increase on the average yield resulted to 5.54 kg ha^{-1} yr^{-1} crab production, this level of production is substantially lower compared with the average yield of 65.4 kg ha^{-1} yr^{-1} obtained by Walton et al. (2006a) in the replanted mangroves in Kalibo, Aklan and those obtained from natural mangroves in Thailand (13 kg ha^{-1} yr^{-1}) and Micronesia (64 kg ha^{-1} yr^{-1}) (Ronnback 1999). This result may reflect the relevance of small-scale releases, such as these stock enhancement trials in augmenting a wild population that is limited by recruitment. However, another factor to be considered which may have caused the low yield in this particular area is the effect of habitat degradation on crab population. The rapid conversion of some natural mangrove areas to *Nypa* plantation may have some harmful effects on the crab population. According to Loneragan et al. (2001), the carrying capacity of a habitat is determined by both environmental and biological factors, such as habitat complexity, availability of food, competition and predation. Langdown (2005), who did a study on burrow occupancy of *Scylla* in the same study site, reported the lack of burrows in areas planted with *Nypa* and their abundance among the complex roots in areas with natural mangroves. Survival and growth rates of penaeid prawns were higher in a high biomass *Enhalus acoroides* seagrass beds

than in a low biomass *Halodule uninervis* area (Loneragan et al. 2001). The presence or total absence of a suitable habitat for release animals may also affect survival in the wild. As in the case of Hamana Lake, Fushimi (1999) stressed that reducing release of hatchery-raised kuruma prawn *Marsupenaeus* (as *Penaeus*) *japonicus* larvae will cause decline in prawn catches because nursery habitats had already been lost. Moreover, Blankenship and Leber (1995) pointed out that a totally degraded area is unsuitable for stock enhancement. Thus, the presence of at least 60% of the original natural mangrove area make the present study site suitable for release and which may have resulted in the relatively high increase in yield coming from released crabs.

Understanding the pattern of growth of individuals is a fundamental requirement for the proper management and conservation of the population (Lee et al. 2006). In this stock enhancement trial, growth was compared between species and between sources of juveniles to identify which species will grow best in this particular area and to determine the best source of stock for future resource management and stock enhancement programmes. Of the three *Scylla* spp., regardless of source, *S. olivacea* showed the fastest growth with *S. serrata* the slowest. *S. tranquebarica* grew at the rate intermediate between the two. The observed growth rates of wild-released *S. olivacea* is comparable to the growth rates for the same species (0.12-0.25 mm d^{-1}) when cultured in ponds as reported by Fortes (1999b). The same was observed in wild *S. olivacea* in India (0.25 mm d^{-1}) (Thomas et al. 1987), Thailand (0.33 mm d^{-1}) (Moser et al. 2002) and the Philippines (0.33 mm d^{-1}) (Walton 2006). However, growth rates obtained for all species regardless of source were lower by at least half compared with the growth rates obtained by Trino et al. (1999) for pond culture of mixed *S. serrata* and *S. tranquebarica* (1.1 mm d^{-1}); Ut (2002) for wild (0.57-0.87 mm d^{-1}) and hatchery-reared (0.56-0.75 mm d^{-1}) *S. paramamosain* grown in either earthen ponds, mangrove ponds or natural mangroves; and Walton et al. (2006b) for wild *S. paramamosain* (0.67 mm d^{-1}). Yet, it is hard to compare growth in the wild with data from ponds, as in pond culture experiments, differences in growth rates may be caused by differences in stocking density, presence or absence of shelters or the kind of diet (Fortes 1999b; Trino et al. 1999; Rodriguez et al. 2003).

In some stock enhancement trials with other species, density and size-at-release were some of the factors that affect growth rate. The growth rate of the tiger prawn

Penaeus semisulcatus in a high biomass seagrass bed for example, declined to lower than average field growth rates at stocking densities >10 m^{-2} (Loneragan et al. 2001). The turbot *Scophthalmus maximus* released at 19 cm length exhibited a growth rate of 0.45 mm d^{-1} (Iglesias and Rodriguez-Ojea 1994) while the ones released at 10-15 cm grew at the rate of 0.10-0.35 mm d^{-1} (Stottrup et al. 1998). Over a 46-wk collection period in Kaneohe and Maunalua Bays, Hawaii, the mean length of hatchery-released mullet *Mugil cephalus* (111.3 and 119.8 mm, respectively) were similar to its wild conspecifics (115.5 and 101.5, respectively) (Leber 1995). The same was observed by Kojima (1995) in the growth of wild and hatchery-reared abalone *Haliotis discus discus* planted in Abu, Tokushima Prefecture. Growth of hatchery-reared abalone corresponded closely to that of its wild conspecifics such that they both reached the legal size of 90 mm shell length at 2+ years old.

In the present study, differences in habitat preferences of the species tested may have caused differences in growth rates. Although *S. serrata* is considered the largest and the fastest growing of the four *Scylla* spp. (Fortes 1999a), the opposite was observed in the present study. Since growth in mud crabs is determined by periodic moulting or ecdysis, the very slow growth rate in *S. serrata* showed that the frequency of moulting may have been less compared with *S. olivacea* and *S. tranquebarica*. This is supported by the observed longer maximum period between release and recovery for non-moulting *S. serrata* individuals compared with that observed in *S. olivacea*.

These three species of *Scylla* which were released in the same area have been exposed to the same environmental factors—temperature, salinity, abundance of food, predators and shelter. However, the fluctuating salinity (ranging 3.8-34.2 p.s.u.) may have affected moulting or growth or even survival in *S. serrata* which prefers a high salinity environment (Hill 1974; Keenan et al. 1998). Although *S. serrata* can tolerate reduced salinity, it is commonly associated with mangrove forests often inundated with full salinity oceanic water and is dominant in oceans where salinity is greater than 34 p.s.u.. On the other hand, *S. tranquebarica* which is often associated with *S. olivacea* is commonly found in mangrove forests and coastlines inundated with reduced salinity seawater (Keenan et al. 1998).

Recent advances on tagging technology provide a basic tool for evaluating stock enhancement success (Blankenship and Leber 1995). The recovery rate of tagged released animals is an indirect measure of their survival in the wild. For example, Leber (1995) evaluated survival rates of released striped mullet from tag recoveries and Davis et al. (2005) also equated tag recovery rates to survival of released *Callinectes sapidus*. In the present study, of the three *Scylla* spp. from different sources, wild-released *S. olivacea* had the highest recovery rates while HR-unconditioned *S. tranquebarica* had the lowest. Moreover, recovery rates of wild-released *S. olivacea* were significantly higher than HR-unconditioned *S. serrata* but did not significantly differ with both HR-conditioned *S. olivacea* and *S. serrata* (ANOVA, $F=6.36$, $p<0.01$). A higher percentage of recovery was also observed for HR-conditioned *S. tranquebarica* compared with the HR-unconditioned ones.

Aquaculture-reared individuals may face some disadvantages upon release into the wild due to differences between natural conditions and the hatchery (Davis et al. 2005) and in most cases hatchery-reared individuals are likely to be less well-equipped behaviourally or morphologically to survive in the wild and the quality of hatchery-reared juveniles may limit the effectiveness of any release programme (Le Vay et al. In press).

In the present study, conditioning hatchery-reared crabs has been shown to improve survival in the wild. It was observed that HR-conditioned crabs had higher recovery and growth rates compared with HR-unconditioned ones, even comparable to wild-released crabs in the case of *S. olivacea*. Crabs conditioned in the pond had been exposed to intraspecific competition, cannibalism, temperature and salinity fluctuations, and had experienced foraging for food. However, crabs kept in individual containers in the hatchery had been provided with filtered seawater with salinity and temperature maintained to constant levels and fed to satiation twice daily.

Anatomical differences between wild and hatchery-reared fish include body shape, abnormal pigmentation, abnormal lateral line development, and opercular, fin ray, branchiostegal membrane and skeletal deformities (Tateishi and Ikeda 1987; Ellis et al. 1997; Hilomen-Garcia 1997; Carillo et al. 2001; Cahu et al. 2003). Hatchery-reared queen conch *Strombus gigas* were found to have thinner shells, shorter spines

and lower survival rates compared with wild conch (Stoner and Davis 1994). Hatchery-reared geoduck clams also showed great variation in shell thickness, resulting in 7-60% shell breakage, depending on the production batch and harvesting method (Velasquez 1992). The ability to avoid predators by hatchery-reared scallop juveniles has also raised concern in Europe (Fleury et al. 1996). Cultured stage IV lobsters failed to develop crusher claws when not given suitable shell materials to manipulate (Govind and Kent 1982). As a result of these defects observed in hatchery-reared organisms, the effect of conditioning has been studied in many species with enhancement potential. Brown and Day (2002) suggested that behaviour in fish being mass-reared for stock enhancement should be conditioned in the same way that captive-bred terrestrial animals and birds for conservation programmes are given the opportunity to learn life skills before release. There is evidence that performance and survivorship of hatchery-reared animals could be improved to a level equating to their wild counterparts by conditioning steps in the hatchery prior to releasing into natural habitats (Delgado et al. 2003; Svasand 2004; Davis et al. 2005). Although most conditioning experiments have been conducted on fish, such conditioning effects are not restricted to vertebrates. Conditioning prior to release has proven to be also effective in increasing fitness of hatchery-reared crustaceans (Wickens 1986; van der Meeren 2001; Davis et al. 2005). According to van der Meeren (2001), shelter-seeking behaviour to avoid predators of the hatchery-reared *Homarus gammarus* has been enhanced through training and experience prior to releasing. Wickens (1986) further showed that provision of hard-shelled prey stimulates normal development of claw morphology in *H. gammarus*. In *Callinectes sapidus*, exposure to predators has caused an increase in their spine length as a defense mechanism (Davis et al. 2005). This increase in spine length directly led to higher survivorship. The findings of the present study and other related studies discussed here showed the importance of conditioning in improving success and efficiency of stock enhancement by alleviating some behavioural and morphological deficiencies in hatchery-reared organisms.

In terms of size classes, recovery rates showed an increasing trend from the smaller to the larger size classes. Not a single crab was recovered from the 20.0-24.9 mm CW released both for *S. serrata* and *S. tranquebarica*. Regression analysis supported this trend showing a positive significant correlation between percent recovery and size

(F=84.04, p<0.001). One of the most important strategies for releasing stocks in stock enhancement programmes is to evaluate the appropriate size-at-release of the organism (Bell et al. 2005). To avoid great loss due to mortality, releasing larger crabs must be considered when dealing with *Scylla* spp. especially in a mixed species situation, since *S. olivacea* were observed to be more aggressive than the other two *Scylla* species (E.T. Quinitio pers. comm.). Antagonistic and cannibalistic behaviour was also observed in *Portunus trituberculatus*, where cannibalism was considered a serious cause of mortality (Okamoto 2004).

Recoveries from releases vary between species, source, size, or the suitability of the release habitat to the species released. Of the 38,773 juvenile snook *Centropomus undecimalis* released in Sarasota, Florida in 1997-2002, only 1,088 (2.8%) were recovered and more than 20% of these recoveries were obtained in the first year with the longest time at liberty of 5.85 years (Brennan et al. 2005). Recovery rates of cultured juvenile Pacific threadfin *Polydactylus sexfilis* (Polynemidae) varied from 0% to 64%, depending on release site (Leber et al. 1998). As summarized by Bell et al. (2005), in penaeid shrimp *Penaeus* (as formerly classified) spp., recovery rates varied from 2.8% to 35.6% depending on the release habitat, size or stage at release, and release season. The present work has revealed the importance of knowing the optimum size-at-release of the target organism. This has been considered by Bell et al. (2005) as one of the most important strategies for releasing stocks in stock enhancement programmes. Larger animals were observed to have better survival than smaller ones which are more vulnerable to predation and cannibalism. In a predation experiment comparing survival of 55-64 mm and 65-75 mm shell height cultured scallops *Pecten maximus* exposed to crabs *Cancer pagurus*, smaller scallops had a mean survival of 25%, while for larger ones 75% (Strand et al. 2004). In *Mugil cephalus*, Leber (1995) and Leber and Arce (1996) reported that recovery rates were directly related to fish size-at-release with 5 times greater recovery rates on fish released larger than 85 mm than those smaller than 60 mm. The optimal size-at-release estimated from a bio-economic model for stock enhancement of *Penaeus esculentus* indicated that releases at 1 g were likely to be more profitable than those at 0.5 g (Ye et al. 2005). In China, releasing shrimps at an early life-history stages soon changed to releasing larger juveniles of ~30 mm total length (Liu 1990). The reason for releasing larger organisms is to lessen size-dependent mortality due to exposure to

predation by larger animals. Giant clams, for example should be reared for approximately 4 years to reach the size when they are no longer vulnerable to predators (Bell et al. 2005). Although size-at-release seemed to be proportional to survival, factors such as costs of production, rearing, and release should be evaluated and its economic feasibility considered as part of the stock enhancement management plan.

Mortality is one of the processes regulating size and structure of populations and both fishing and natural mortalities are parameters required for successful management of exploited stocks (Brey 1999). Mortality rates in wild-released *S. olivacea* were higher than in both HR-conditioned *S. serrata* and *S. tranquebarica*. Partitioning total mortality rate into fishing and natural mortalities showed that 76% of mortality in *S. olivacea* was due to fishing. The high fishing mortality of *S. olivacea* is reflected by its high recovery rate. In the case of *S. serrata*, the relatively low total mortality had a higher component of natural mortality. Differences in mortality rates may be accounted for by the differences in behaviour of these species. *S. serrata* is considered as free-ranging, compared with *S. olivacea* which prefer to live in burrows (Estampador 1949; Fortes 1999b), and may not be as attracted to traps as the latter since they are highly mobile and may feed throughout the tidal cycle. *S. serrata* may also be more dispersed in the mangroves as shown by the longer interval of recovery from the day of release to the day when the last tagged crab was recovered. *S. olivacea*, on the other hand, tends to live in intertidal burrows and thus feed at the onset of high tide and may easily be more attracted to baited traps because of the longer interval between feedings. The deployment of traps which starts prior to the onset of high tide and retrieved on the next low tide may also target *S. olivacea* that have just come out of their burrows and searching for food. The high recoveries during sampling following release resulted in a shorter interval from release to recovery of the last tagged crab since many of them have been caught on the first or second sampling. This trend of higher recoveries following release was also observed by Leber et al. (1995) and Brennan et al. (2005). Considering the more free-ranging behaviour of *S. serrata*, and hence lower attraction to traps, mortality rates may be slightly underestimated. Since survival is based on recovery rates, this low recapture rate may have slightly underestimated survival for this species. *S. olivacea* which tend to be caught more easily may have over estimated mortality rates. Total mortality (per

month) obtained for any of the released species in this particular study is comparable to the estimated mortality for the wild *Scylla* spp. population obtained during the mark-recapture study in the same area which ranged from 0.14 to 0.45 per spring tide (~per half month) (see Chapter 4). These values are also comparable to the total mortality rates obtained by Ut (2002) for *S. paramamosain* in Vietnam (0.84 month^{-1}, =0.42 per half month), and by Walton (2006) for *S. olivacea* in New Buswang, Aklan (33-45% between samplings, ~per half month). Using CPUE data from *S. serrata* population in a South African estuary where there was no fishing activity and a sand bar hinders population exchange, Hill (1975) estimated an annual natural mortality of 41 and 60% for two succeeding years. This revealed that released crabs have almost the same survival as their wild conspecifics in the natural environment. This finding is important since according to Stoner and Davis (1994), stock enhancement will only be effective if hatchery-reared and wild stocks are identical in growth and mortality.

If survival is proportional to tag recovery rates (Leber 1995; Davis et al. 2005), then hatchery-reared animals tend to have higher mortality than their wild conspecifics. Aside from the result of the present study showing higher recovery rates in wild-released *S. olivacea* compared with the hatchery-reared ones, this observation had also been reported in other species. Recovery of tagged wild mullet *Mugil cephalus* released in Kahaluu stream, Hawaii was higher (4.9%) than its hatchery-reared conspecifics (2.1%) (Leber et al. 1995). In a study conducted by Davis et al. (2005) comparing recovery rates of wild and hatchery-reared blue crab *Callinectes sapidus* released in Aberdeen Cove of the Chesapeake Bay, recovery rates over a 60-d period were also significantly higher in wild (2.69%) than in hatchery-reared (1.12%) crabs.

The results of the present work showed that, generally, *Scylla* spp. tended to remain close to where they were released (Fig. 5.15). The maximum distance travelled by the crabs caught farthest from the release site was estimated to be 1 km for those released in Site A and caught in collection Area 5, and 1.4 km for those released in Site B and caught in the other end of collection Area 2. In mark-recapture experiments conducted by Walton (2006) in Panay Island, Philippines, *S. olivacea* were reported to have limited movement and were presumed to have probably foraged around their burrows. In another mark-release-recapture study of *S. serrata* in South Africa, Hill (1975) also observed limited movement. Despite their capability to move along the estuary by at

least 800 m in one night, *S. serrata* preferred to remain in a restricted area (Hill 1978). Moreover, in a study conducted in Moreton Bay, Australia, Hyland et al. (1984) reported that *S. serrata* in a narrow creek with mangrove-covered banks displayed little movement. Those crabs with direct access to the sea were caught within the 1 km distance from the release site until 36 weeks after release. In a mark-recapture study of *S. paramamosain* in an island mangrove in Vietnam, although 2 crabs were recovered 12 km away from the release site, Ut (2002) suggested that majority of the crabs remained close to the study area.

Noticeably, most of the *S. serrata* released in Sites B and C moved to collection Area 3 where most of them where recovered. *S. serrata* is considered highly free-ranging while *S. olivacea* tends to live in burrows (Estampador 1949; Fortes 1999b). If *S. serrata* do not burrow as much as *S. olivacea*, movement to collection Area 3 from release Site C may have been caused by the absence of structures that could provide them shelter and protection from predators in the release site or its vicinity. Site C and its neighbouring collection areas is mostly *Nypa fruticans* plantation (see Chapter 2) and not much of the complex mangrove root system can be found in the area. However, collection Area 3 which is closer to release Site B, with the different species of mangroves in its surroundings, can provide more shelter to the nomadic *S. serrata*. *S. olivacea* being hole-dwellers can seek refuge in their burrows in the absence of shelter provided for by the mangroves. Moreover, competitive interactions may have caused *S. serrata* to evade *S. olivacea* which is known to be the more aggressive species (E.T. Quinitio pers. comm.).

In general, movement of crabs within the mangroves may be caused by scarcity of food, avoidance to predators, or finding an appropriate refuge. In a study of the feeding behaviour of *S. serrata*, Hill (1979a) observed that they moved continuously until they locate prey animals. Crabs were reported to occur more frequently in areas where they have the highest number of prey organisms (Hill 1979a). Results of the mark-recapture study of wild population in Chapter 4 also showed minimal movement of crabs from release sites to recapture areas. Released crabs in the present work followed the same trend showing minimal displacement. If food is one of the major factors that determines crab movement then it can be assumed that there is enough food in the Ibajay mangroves not just for the wild mud crab population but also for

the newly added released ones. According to Bell et al. (2005), the shallow, inshore habitat of many invertebrate species (including *Scylla* spp.) with potential for stock enhancement, combined with their lack or limited mobility facilitates assessments of the success of the interventions and aids adaptive management.

Fidelity to release sites or limited post-tagging movement has been observed not only in the present study and other studies of *Scylla* spp. (Hill 1975; Hill 1978; Hyland et al. 1984; Ut 2002), but also in other species. Tagged juvenile blue crabs *Callinectes sapidus* (35-62 mm CW) released in two tidal marsh creeks in North Carolina showed an extremely low emigration rate of 0.02 crabs d^{-1} in one creek and 0 crab d^{-1} in the other creek, with a mean total distance moved of 25 and 19 m d^{-1}, respectively, indicating fidelity of these crabs to the tidal marsh (Eggleston et al. 2004). Released lobster *Homarus gammarus* also contributed as high as 35% to the catch within their release areas because most of them did not stray (Bannister et al. 1994). Cultured striped mullet *Mugil cephalus* showed a strong tendency to remain in the vicinity of release sites, regardless of release season or size-at release (Leber et al. 1995; Leber and Arce 1996; Leber et al. 1997).

Evaluation of the results of the stock enhancement trials

SPECIES SELECTION – The findings of the present study revealed the importance of understanding the habitat and the population dynamics of the different target species of interest in choosing the most suitable species for stocking. This particular site, for example, where salinity sometimes drop to 3 p.s.u. during the rainy season may not be suitable for *S. serrata* which prefer full salinity oceanic water but is best fitted for *S. olivacea* or *S. tranquebarica* which prefers reduced salinity seawater (Keenan et al. 1998). Preference to high salinity of *S. serrata* was also reported by Hill (1974). The result showing that *S. olivacea* had the highest recovery rate and growth rates are confirmations that indeed this particular habitat is most suitable for *S. olivacea* (see Chapter 3). Based on growth rates of the three *Scylla* species, it can be concluded that the ideal species to be used for stock enhancement in this particular area is *S. olivacea*. The results of the study also revealed that both wild-released and wild (see Chapter 4) *S. olivacea* had growth rates of 0.25 mm d^{-1} which are significantly lower by 0.14 mm d^{-1} than its hatchery-reared, pond-conditioned

conspecifics (ANOVA, $F=29.79$, $p<0.001$). These findings may promote the utilization of hatchery-reared crabs and would spare wild crabs from other less depleted areas of being collected and released in an overfished area. This makes it more promising for stock enhancement programmes because crabs can then be purely obtained from the hatchery and dependence on wild source will be at least minimized or even eliminated. Ut (2002) conducted two experiments on growth of wild and hatchery-reared *S. paramamosain* which got contradicting results. In the first experiment where wild and hatchery-reared conspecifics were stocked together in ponds without shelter, a significantly higher growth rate and survival was observed in wild crabs than hatchery-reared animals. However, in the second experiment where shelters (plastic pipes and bunches of sticks) were provided, hatchery-reared crabs had significantly higher growth rates. These differences were explained by Ut (2002) as a result of environmental differences, differences in nursery culture, or batch to batch variations in both wild and hatchery-reared crabs. Identical growth of hatchery-reared and wild stocks is important in order for stock enhancement to be effective (Stoner and Davis 1994). Kellison et al. (2000) also stressed that hatchery-reared organisms should be able to survive and perform as well as the wild conspecifics. The result of the present study even showed a better growth rate for hatchery-reared, pond-conditioned *S. olivacea* than for both wild-released and the natural population in the study area. This reveals that hatchery-produced juveniles once conditioned can cohabit with the wild ones and survive to compete effectively for resources.

CONCLUSION

This chapter presents the results obtained after several batches of the three species of *Scylla* from different sources have been released in the natural mangroves of Naisud and Bugtong Bato, Ibajay, Aklan, Philippines. This is the first stock enhancement trial on the mud crabs *Scylla* spp. involving the use of coded microwire tags as a tool for determining release success. The results of this study document that small-scale releases such as this stock enhancement trials can increase abundance of mud crabs in a partly isolated marine habitat such as the present mangrove study site. Both growth rates and survival rates suggest that *S. olivacea* is the best suited species for the

mangroves of Naisud and Bugtong Bato, Ibajay, Aklan. The results further reveal that pond-conditioned *S. olivacea* can have higher growth rates and recovery rates equivalent to those of their wild conspecifics. This is a very promising finding for future resource management and stock enhancement programmes since *S. olivacea* needed for release can be purely obtained from the hatchery so long as they are conditioned prior to release. Aside from conditioning hatchery-reared juveniles, size-at-release of mud crabs is also an important consideration. This experiment showed that crabs measuring 49.9-79.9 mm CW had significantly higher recovery rates than the ones measuring <45 mm CW with the optimum size-at-release of 65.0-69.9 mm CW regardless of species or source. It was observed that, generally, *Scylla* spp. tended to stay where they were released or just moved around the vicinity of the release site. Movement of crabs within the mangroves may be determined by availability of food or presence of predators. According to Hill (1979a), they tend to occur more frequently in areas where they have the highest number of prey organisms. Considering food as a factor that influence crab movement, it can be assumed that food is abundant in this particular study area since crabs tend to stay in areas where they have been released.

According to Bell and Gervis (1999), candidates for restocking and stock enhancement in the Pacific islands are those inshore species of high value that have been overfished and/or limited by recruitment. And once the species in question and management goals are determined, the ten essential components of a "responsible" enhancement programme can be distilled into three critical issues: 1) understanding the nature of the system; 2) producing robust, compatible individuals for release; and 3) evaluating the effects of releases (Blaylock et al. 2000). The information obtained from pilot studies such as this stock enhancement trial is important in modifying release protocols to improve recovery rates. And based on the results of this stock enhancement trial, *S. olivacea* showed to be the most appropriate species for release in this particular mangrove area. Release strategies such as conditioning hatchery-reared juveniles in the ponds for at least one month and growing them to a minimum size of at least 49 mm CW prior to releasing them to the wild proved to be essential. These release strategies have been demonstrated to be effective in obtaining higher recovery and growth rates. These findings may be useful in the future resource management and stock enhancement projects of mud crabs. Information from this

pilot stock enhancement trial may be revised to further improve the effect of hatchery-produced releases in the wild. Moreover, future endeavours should carefully follow the components for responsible enhancement programmes in order to minimize impact of released organisms on the wild population. From the three critical issues presented by Blaylock et al. (2000), focus should be in the evaluation of the effects of releases.

CHAPTER 6

Summary

SUMMARY

Marine ecosystems are one of the most important resources of the Philippines, an archipelago bounded by 17,460 km of coastline and 26.6 million ha of coastal waters (Primavera 2000). Mangroves are one of those marine resources that play an important economic and ecological role, but have been severely damaged due to anthropogenic activities (Kuhlmann 2002). Loss of mangroves means loss of habitat to many marine species that use the habitat both as nursery or permanent residence. Many studies have shown that nearshore catches of shrimps, shellfish and finfish are positively correlated with mangrove areas (Macnae 1974; Staples et al. 1985; Camacho and Bagarinao 1986; de Graaf and Xuan 1998; Kathiresan and Rajendran 2002; Manson et al. 2005). Hence, loss of habitat has led to declining populations of most of these marine species. This further aggravates overexploitation of these fisheries resources. Declining marine stocks is not a recent trend; it dates back to as early as 1880 in the North Sea (Gulland and Carroz 1968). In the Philippines, capture records for fisheries of mud crabs *Scylla* spp., an important resource coming from the mangroves, started to decline in 1992 (FAO 2006). Aside from the FAO data, most fishermen have observed that mud crab catches have been decreasing through the years. This problem can be addressed through regulation of fishing effort, rehabilitation of mangrove habitats, mangrove-friendly aquaculture, and enhancement of wild stocks (Heasman and Fielder 1977; Jayamanne 1992; Robertson and Kruger 1994; Blankenship and Leber 1995). The present work attempts to demonstrate the potential for addressing the problem through stock enhancement. Prior to any release of stock for the purpose of enhancing depleted wild stocks, there is a need to study the target habitat and obtain baseline information on the population of the species concerned (Blankenship and Leber 1995; Bell et al. 2005).

THE MANGROVE HABITAT – Results of the community structure analysis of the mangroves in Naisud and Bugtong Bato, Ibajay, Aklan showed a highly diverse mangrove area, with 27 species of true mangroves. However, human activities such as pond construction, *Nypa fruticans* plantation, mangrove cutting, were very evident and pose a great threat to the high species richness of the area. Almost 30% of the once pure stands of natural mangrove area have been lost to these activities. Of the

70-ha area surveyed, approximately 50 ha remains as natural mangroves that is suitable as mud crab habitat. The mangrove area has a narrow access to the sea which may limit recruitment, and the combined effect of limitations on recruitment and overexploitation may explain the low mud crab yield in the area. Limited recruitment coupled with a partially degraded mangrove habitat, makes the area ideal for stock enhancement trials. Moreover, the restricted access to the sea may limit emigration of released crabs, hence, more crabs may be available for the fishery.

POPULATION AND FISHERIES OF *SCYLLA* – Following the recent re-classification of the genus *Scylla* (Keenan et al. 1998; Imai et al. 2004), monthly monitoring of crab landings revealed the sympatric occurrence of three species of mud crabs in the study area; *S. olivacea*, *S. serrata* and *S. tranquebarica*, with the predominance of *S. olivacea*, comprising 95% of the total catches. The pattern of differences in habitat preferences among different sizes of mud crabs had been reported in Australia for *S. serrata* (Hill et al. 1982) and in Vietnam for *S. paramamosain* (Walton et al. 2006b). However, these preferences were not observed in *S. olivacea*, in the present work. Instead, a high degree of fidelity to the intertidal mangrove habitat was revealed; except for the absence of berried crabs, all sizes and stages were caught inside the mangroves. The absence of berried crabs inside the mangroves and their occasional presence near the mouth of the river may suggest offshore migration for spawning, a naturally occurring event also reported by Arriola (1940), Hill (1975), Hyland et al. (1984), Robertson and Kruger (1994) and Moser et al. (2005). The consistently high numbers of immature female crabs throughout the sampling period and the lack of modal progression in size-frequency distributions may indicate year-round recruitment, which however, due to the limited access to the sea produced low mud crab yields. However, changes in mean crab size indicate possible periods of higher abundance of smaller newly-recruited crabs. The low mud crab yield and the negative significant correlation of CPUE in terms of biomass with time coupled by the decreasing mean CW and BW through time may indicate growth overfishing of the population being studied. Overall, these results present an interesting opportunity for stock enhancement trials.

ABUNDANCE AND GROWTH RATES OF *SCYLLA* – Knowledge of the crab abundance in the habitat is necessary prior to any stock enhancement work.

Population, growth rate and movement of the crabs within the mangrove area were estimated through mark-recapture methods and analysed using the Jolly-Seber model for open populations. This model gives estimates of population parameters such as abundance, survival/mortality and recruitment. Mark-recapture methods for estimating population had been widely used in *Scylla* spp. (Hill 1975; Williams and Hill 1982; Robertson and Piper 1991; Le Vay et al. 1999; Moser et al. 2002; Ut 2002; Walton 2006). Population estimates revealed a very low density of crabs ranging from 14 to 34 crab ha^{-1} comparable to the density obtained for a heavily fished area in Ranong, Thailand (Moser et al. 2005). This has been supported by CPUE data obtained on the same period which paralleled the population estimates during each sampling. The growth rates of crabs which were 0.25 mm CW d^{-1} and 1 g BW d^{-1} are almost the same as the untagged *S. olivacea* both in the wild (Thomas et al. 1987) and in pond culture (Fortes 1999b). This suggests that the coded microwire tags did not affect growth of *Scylla* spp. Moreover, it was observed that generally, crabs tended to stay where they were released or just moved around the vicinity of the release site. According to Hill (1979), movement of crabs within the mangroves may be determined by availability of food and they tend to occur more frequently in areas where they have the highest number of prey organisms. It can therefore be assumed that food is abundant in the study area.

The results of the studies combined together support the need for and the suitability of stock enhancement as an approach to increasing mud crab production in this particular mangrove area. These are the partially degraded mangrove habitat with 50 ha suitable for crabs, limited recruitment, low mud crab yield, decreasing CPUE, mean CW and BW, low density, acceptability of CWT to crabs, and limited movement of crabs which may indicate sufficient food in the area.

STOCK ENHANCEMENT TRIALS – Small-scale releases, such as the stock enhancement trials reported in Chapter 5, have increased abundance of mud crabs in a partly isolated marine habitat. Results suggested that *S. olivacea* is the best suited species for this particular habitat as revealed by the higher growth rates and survival of both the wild-released and hatchery-reared pond conditioned crabs compared with the other *Scylla* species released. The results also revealed that conditioning hatchery-reared juveniles prior to release may improve survival in the wild and may be

essential in future enhancement programmes. Moreover, regardless of species or source, size-at-release is also an important factor for survival, such that crabs measuring 49.9-79.9 mm CW had significantly higher recovery rates than the ones measuring <45 mm CW. This however, should be balanced with the cost of production in growing crabs to desired size-at-release. The limited movement of wild crabs as shown by the results of the previous mark-recapture study was corroborated in the present study showing that both wild-released and hatchery-reared crabs exhibited minimal movement within the mangroves. This may suggest that food in the area is enough to support both the wild population and the added released crabs.

GENERAL CONSIDERATIONS – The result of this stock enhancement trial will be important in designing or planning future stock enhancement or resource management programmes not just in this particular study area but in any *Scylla* spp. habitat in general. Tagging proved to be an important tool in providing quantitative basis for assessing stock enhancement success by enabling monitoring of recoveries from the released stock.

Based on the definitions given for stock enhancement, restocking and sea ranching, restocking is perhaps the most appropriate description of the approach used in addressing the problems of decreasing mud crab population in the study area. According to Bell et al. (2005), restocking should be considered when the time required for replenishment of the population by natural processes, under management measures such as a moratorium on fishing or the use of marine protected areas, is likely to be unacceptable to the fishing community. Imposing a moratorium on fishing may be difficult in communities that mainly depend on fishing for livelihood, as is the case in most of the coastal areas in the Philippines. Supplementary livelihoods must be considered as part of the stock enhancement programme. Proper information dissemination, especially to the local government and the community involved, is suggested to be considered prior to any resource management or stock enhancement endeavour. Success of projects of this kind partly depends on the participation and cooperation of the community who are the principal recipients or the beneficiary of the project. Fernandez et al. (2000) cited some key conditions for a successful management and development of large marine ecosystems such as the delineation and congruence of politically defined space with ecological realities; enhancement of

stakeholder management capabilities (i.e., information, resources, and skills); supplementary livelihood opportunities; enabling institutions and increased grassroots participation; and effective oversight and coordination between and among concerned state, civil society, and market (profit-oriented) organisations.

If restocking aims at rebuilding spawning biomass of severely depleted populations to levels where the fishery can once again support regular harvest, then the result of the present study rather fits the concept of sea ranching, where animals are released to grow and later on are harvested for food fisheries. The short period between release and recovery of a higher percentage of *S. olivacea* may not be enough to build the spawning biomass of the population but significant enough to increase yield and production. However, in the future, when a practical stock enhancement programme is implemented, this may change since the need to conduct regular monthly sampling may be eliminated and monitoring could be started for quite some time after release, thus giving the animals enough time to grow, disperse, reproduce and interact with their wild conspecifics.

If carefully applied and properly managed, stock enhancement technology has considerable potential as a fishery management tool. However, the results presented here after several batches of releases are a first step in information gathering. For future projects, these need to be coupled with additional management considerations to provide a controlled approach to marine enhancement such as development of a species management plan, well-defined indicators of success, prevention of genetic inbreeding and outbreeding depression, disease and health management, consideration of ecological interactions, identification of socioeconomic realities and use of adaptive management strategy (Leber et al. 1995).

Stock enhancement presents hope to the declining fisheries production which has been a problem confronting developing countries such as the Philippines. Food security and sustainability has been the common goal of the government, non governmental organisations and research institutions. Adapting and proper implementation and management of stock enhancement may increase production and may help achieve sustainability and food security.

The results presented and the strategies applied in this stock enhancement trial may be useful in future endeavours. However, this may apply only to such isolated habitats, such as the study area. A different scenario may be expected in a totally open population where recruitment is not limited and crabs have more access to migrate to some other suitable neighbouring habitats. Other approaches in addressing the declining fisheries resources should also be considered. Habitat restoration, for example, has been proven successful in increasing mud crab yield in the replanted mangroves in Kalibo, Aklan (Walton 2006a). The need for gathering baseline information about the habitat and the species concerned is always basic in order to identify the most suitable approach in dealing with the problem.

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