COSTS AND BENEFITS OF CHANGING SAMPLING METHODS IN LONG-TERM MONITORING OF MARINE PROTECTED AREAS

by

Iwao Fujii

B.Sc, Tokyo University of Marine Science and Technology, 2014

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF

THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

in

THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

(Zoology)

THE UNIVERSITY OF BRITISH COLUMBIA

(Vancouver)

June 2018

The following individuals certify that they have read, and recommend to the Faculty of Graduate and Postdoctoral Studies for acceptance, a thesis / dissertation entitled:

Costs and benefits of changing sampling methods in long-term monitoring of marine protected areas

submitted by Iwao Fujii in partial fulfilment of the requirements for

the degree of Master of Science

in Zoology

Examining Committee:

Amanda Vincent, Zoology
Supervisor

Mary O'Connor, Zoology
Supervisory Committee Member

Christopher Harley, Zoology
Supervisory Committee Member

Eric Taylor
Additional Examiner, Zoology

Abstract

To confirm their effectiveness, long-term monitoring of marine protected areas (MPAs) is critical. Such longitudinal sampling can, however, pose problems of methodology, particularly as more preferred tools may evolve over time. My research sought to understand how two commonly used methods, line intercept transect (LIT) and photoquadrat (PQ), which were used sequentially in MPA monitoring, estimated percent coral cover in a series of MPAs in the central Philippines.

It can be difficult to reconcile data from LIT and PQ because their different parameters and sample sizes can generate contradicting results. I specifically addressed the issues of (i) small sample sizes of LIT data because of a challenge in maintaining fixed transects and (ii) possible spatial autocorrelation of PQ data owing to a decision to place transects close together. The data also faced other problems, including numerous missing data points and inconsistent data labeling.

Analyzing three years of data when both LIT and PQ were applied, I found that LIT estimated higher mean percent coral cover than PQ. However, subsampling LIT data – and thus increasing sample sizes – improved comparison of estimates by these two methods. This finding demonstrated that PQ data, with larger sample sizes from wider spatial ranges, detected significant temporal patterns of coral cover whereas LIT data did not. This result confirmed that more observations and a wider spatial extent of sampling increase the power to detect statistical significance. I also found that PQ data did not exhibit significant spatial autocorrelation, eliminating one of concerns in analyses.

Two other aspects of monitoring also stand out in their importance for better assessment of coral cover. The first is the need for frequent communication between field biologists and researchers / analysts to assure the proper execution of sampling procedures and reporting of any changes to field methods. The second is the vital need to minimize errors by ensuring the accuracy of data encoding and use of standardized labeling systems over time. The quality of the data determines the accuracy

of coral cover estimates, especially when different methods are used in MPA monitoring.

Lay Summary

Tracking how coral responds to marine protected areas is difficult. This is particularly true when sampling methods change part way through long-term monitoring. My research explored how to combine data from the two sampling methods: line intercept transect (LIT) and photoquadrat (PQ). In LIT, we lay and follow underwater measuring lines, recording distances where the bottom cover changes. In PQ, we take digital photographs of square areas along underwater lines then sample points within those images, recording what we observe. In this study, I found that LIT estimated higher coral cover than PQ. However, the estimates became more similar after I separated the large amount of LIT data into smaller subsets, which created more replicates of data and made analyses statistically more powerful. I also found that we were more likely to detect significant coral cover change over time if we collected more observations and sampled across larger areas.

Preface

I developed the research questions and methodological design of this thesis collaboratively with Dr. Amanda Vincent and Dr. Jennifer Selgrath. My committee members, Dr. Christopher Harley and Dr. Mary O'Connor, also gave me wise advice on how best to generate a coherent thesis. Data used for Chapter 2 of this thesis was primarily collected by Project Seahorse Foundation for Marine Conservation (currently Zoological Society of London - Philippines) between 2008 and 2010. However, the entire dataset comprised MPA monitoring between 1998 and 2016, and I took part in the monitoring survey in Spring 2016 to obtain a new set of data for this year. These data were collected to observe temporal patterns of coral cover in marine protected areas located in Danajon Bank, Bohol, the Philippines. I carried out all analyses and prepared all manuscripts in this thesis; however, Dr. Jennifer Selgrath made notable contributions to data manipulations and analyses. Chapter 2 is in preparation for publication. I am the lead author, along with Dr. Amanda Vincent and Dr. Jennifer Selgrath.

Table of Contents

List of Tables

List of Figures

List of Symbols

LL – Line intercept transect line data (before subsampling)

LP – Line intercept transect point data (after subsampling)

P230 – PQ data with all number of quadrats (sample n = 230)

P40 – PQ data with 40 randomly selected quadrats

LL+P230 – LIT line and PQ 230 combined data

LP+P40 – LIT point and PQ 40 combined data

List of Abbreviations

CBD – Convention on Biological Diversity

CPCe – Coral Point Count with Excel extensions

IUCN – International Union for Conservation Nature

LIT line + PQ 230 – LIT line and PQ 230 combined data

LIT line data – Line intercept transect line data (before subsampling)

LIT point + PQ 40 – LIT point and PQ 40 combined data

LIT point data – Line intercept transect point data (after subsampling)

LIT – Line Intercept Transect

LPT – Line Point Transect

ML – Maximum Likelihood

MPA – Marine Protected Area

PQ 230 data – Photoquadrat data with all number of quadrats (sample n = 230)

PQ 80 data – Photoquadrat data with 80 randomly selected quadrats

PQ 40 data – Photoquadrat data with 40 randomly selected quadrats

PQ – Photoquadrat

PSF – Project Seahorse Foundation for Marine Conservation

REML – Restricted Maximum Likelihood

ROV – Remote Operated Vehicle

SCUBA – Self-contained Underwater Breathing Apparatus

UBC – University of British Columbia

ZSL – Zoological Society of London

Acknowledgement

Foremost, I would like to express my sincere gratitude to my supervisor, Dr. Amanda Vincent, for her continuous support to my MSc study. Her guidance helped me in all the time of research, and led me to a deep perspective of marine conservation. Besides my supervisor, I would like to thank my thesis committee, Dr. Mary O'Connor and Dr. Christopher Harley, and my examiner, Dr. Eric Taylor for their insightful comments and questions. My special thanks also goes to Dr. Jennifer Selgrath, who generously offered me her statistic knowledge to tackle challenging analyses and provided precise advice to improve my thesis. I could not have completed my research without their contribution.

My MSc study was supported by Ito Foundation for International Education Exchange, the Faculty of Graduate and Postdoctoral Studies of the University of British Columbia, Project Seahorse, and Zoological Society of London. I especially would like to acknowledge staff members at Ito Foundation for their support. They cheered me up when I faced difficulties through warm messages from the other side of the Pacific Ocean, and connected me with many other fellows in the world, who motivated me to work even harder by showing me their immense passion for their own field.

I was very fortunate to have an opportunity of being part of long-term monitoring survey on MPAs in the Philippines. I could not have had such an invaluable experience without generous support by Angelie Nellas, who hosted my visit, and Myrtle Arias, Nicholson Tan, and Jofel Coching, who led the long-term monitoring survey, and all other staff members in Zoological Society of London - Philippines. I also would like to thank Xyza Billones, Mar John Piencenaves, Romel Anasco, and all other survey members for making my time in the Philippines wonderful.

I am very grateful to all colleagues in my lab, Project Seahorse: Dr. Sarah Foster for generously sharing her knowledge and skills to handle my complicated dataset, Scott Finestone for his prompt, flexible action to logistically support my field season and study, Regina Bestbier, Riley Pollom, and

Lily Stanton for providing me with tips to further improve my research. Thank you to Kyle Gillespie for generously contributing his time to checking my long thesis drafts, Dr. Ting-Chun Kuo for sharing her wonderful statistical knowledge with me, Xiong Zhang for giving me a full set of critical comments on my draft, Tanvi Vaidyanathan for her generous help whenever I got in trouble, and Clayton Manning for cheering me up when I had difficulties in the lab. I particularly would like to offer my thanks to Emilie Stump for being the awesome buddy both in the lab and underwater.

I could not have survived tough FISH 520 classes without wonderful supportive peers; I was very fortune to have Santiago de la Puente, Madeline Cashion, Muhammed Oyinlola, Tayler Clarke, Lian Kwong, Abdulrahman Ben Hasan, Travis Tai, Natalie Mahara, and Maureen Finn in the same cohort both inside and outside classes. Thank you as well to Emily Adamczyk for being a nice diving and hiking buddy, and every single colleague I met in Institute for the Oceans and Fisheries and Department of Zoology for inspiring me by sharing wonderful thoughts, knowledge, and skills.

I could come to study in Vancouver thanks to Dr. Jesse Ronquillo for his persistence to connect me with Dr. Vincent. Dr. Alicia Ronquillo, Jesse's wife, always made me feel Vancouver my home away from home. I also would like to thank Mr. Norihiko Higuchi and his wife, Mrs. Kayoko Higuchi for warmly welcoming me to Vancouver, and again made me feel this place my home.

My acknowledgement cannot be completed without mentioning tremendous support from my family: my mother, Cynthia for always caring about my health and everything while I was away; my father, Tomoyuki for always encouraging me to embark on new challenges; my sister, Kei for always updating me on how my family was. Thank you to my grandparents in the Philippines, Adelaida and Felix, for making me proud of being half Pinoy, and Uncle Charlie and Aunt Mila in San Francisco for being my parents in North America. Lastly, I would like to express my thank to my grandfather in Japan, Yorio, who passed away in 2013, for always being on my side in my heart.

Dedication

To my family who gave me strength to make it through all challenges I encountered

A truly brave man is ever serene; he is never taken by surprise; nothing ruffles the equanimity of his spirit. In the heat of battle he remains cool; in the midst of catastrophes he keeps level his mind. Earthquakes do not shake him, he laughs at storms. We admire him as truly great, who, in the menacing presence of danger or death, retains his self-possession; who, for instance, can compose a poem under impending peril or hum a strain in the face of death.

– Inazo Nitobe, Bushido: The Soul of Japan

Chapter 1: Introduction

1.1 Rationale

Marine protected areas (MPAs) are common measures to protect coral reef habitats from fishing impacts. Assessing the long-term protection effect on habitats is important for effective MPA management, but challenging. This is especially the case when sampling techniques are changed midway through long-term monitoring. The goal of this study was to compare estimates from different sampling methods in terms of mean percent coral cover and temporal patterns of percent coral cover. This study also discussed the costs and benefits of changing methods in assessing habitat protection.

1.2 Background

1.2.1 Coral reef habitats essential, but threatened

Coral reefs are critical marine ecosystems in supporting high biodiversity. Being mainly located in warm shallow waters, estimates suggest that coral covers up to 0.5 % of the world ocean floor (Smith, 1978; Copper, 1994; Spalding and Grenfell, 1997; Spalding et al., 2001). Nevertheless, coral reefs are the home to nearly one-third of the marine fish species in the world (McAllister, 1991; Wilkinson and Souter, 2008), and estimates of the number of species dwelling on coral reefs range from 50,000 to nearly 10 million (Knowlton et al., 2010). Coral reefs provide food and shelter to numerous organisms, and provide them with important spawning and breeding areas (Moberg and Folke, 1999). Such functions are enabled by scleractinian corals, also known as hard coral, that create the complex three-dimensional structure of coral reefs. This three-dimensional structure allows for niche diversification and high levels of species and functional biodiversity (Paulay, 1997).

Coral reefs benefit vast numbers of people by providing goods and services including coastal protection, recreational opportunities, and cultural development (Smith, 1978; Kuhlmann, 1988; Spurgeon, 1992; Done et al., 1996; Peterson and Lubchenco, 1997). They also support numerous different types of fisheries (UNEP-WCMC, 2006), which provide a source of food for millions of people who depend on the reef environment (Cinner et al., 2013). Approximately 75 % of the world's coral reefs occur in developing countries (Pauly et al., 2002), and these countries are driving much of the global demand for, and catch of, fishery resources owing to their rapidly growing populations (Merino et al., 2012). Furthermore, it is estimated that developing countries will account for more than 91 % of the world's total increase in fish consumptions by 2030 (Béné et al., 2015). Given the importance of coral reefs in supporting both marine ecosystems and people's livelihoods, maintaining the quality of reefs is of high priority to sustain benefits we receive from the reef environment.

Despite the importance of coral reefs for both biodiversity and humans, they face a number of threats (Jackson et al., 2001). Fishing puts pressure on many habitats including coral reefs (Schleyer and Tomalin, 2000; Yoshikawa and Asoh, 2004; Mangi and Roberts, 2006). Trawling, in which a fishing net is towed over the bottom of the sea, is a cause of widespread damage to coral communities on tropical shelves (Wilkinson and Slavat, 2012). The impact of fishing with explosives and sodium cyanide is also one of the major disturbances of marine ecosystems in tropical regions such as the Philippines and Indonesia (Mora et al., 2011; Hughes et al., 2013). Such destructive fishing methods reduce the complexity and diversity of habitats, which result in decreases in fish biomass and species richness (Kaiser et al., 2002; Gratwicke and Speight, 2005). In addition, overfishing of herbivore fishes has led to a profound phase shift from the coral-dominant to algae-abundant environment in many places (McManus and Poslsenberg, 2004; Bellwood et al., 2011). These fishes normally prevent algae from over-growing coral and facilitate coral recruitment by grazing algae on reef rocks; thus, their removal poses problems (Mumby et al., 2006; Mumby and Steneck, 2008).

Threats such as terrestrial agriculture, deforestation, and coastal development are introducing sediment, nutrients, and other pollutants into coastal waters (Schaffelke et al., 2012; Kroon et al., 2014). The resulting turbidity and algal growth can cover coral or cause widespread eutrophication, leading to degradation of the reefs' productivity or even death (Fabricius, 2005). Coral reefs are also susceptible to carbon emissions, and associated increases in sea surface temperature and acidification, which often result in severe degradation of the habitat quality (Pandolfi et al., 2011; McCulloch et al., 2012). Large and severe devastation of coral reefs resulting from these threats provoke calls for immediate conservation action to ensure their healthy conditions (Bridge et al., 2013).

1.2.2 MPAs and habitat protection

MPAs are becoming a mainstream management tool for marine conservation (Juffe-Bignoli et al., 2014). Of particular importance are no-take marine reserves, where fisheries are prohibited, which are often established with an objective of sustaining or enhancing fisheries and curbing habitat degradation (Walmsley and White, 2003; Sale et al., 2005). The assessment by the protected planet in 2017 revealed that approximately 6.96 % of the global ocean area was protected by MPAs (MPAs include only those that meet the definitions of protected areas by the International Union for Conservation of Nature: IUCN, and the Convention on Biological Diversity: CBD). However, CBD called for 10 % of marine areas to be protected by some form of MPAs by 2020 in Aichi Target 11 set in 2010 (Spalding et al., 2012). The same target was adopted as part of the 14th goal of the Sustainable Development Goals in 2015 (UN, 2015). Scientific recommendations for the spatial extent of MPAs are greater than the target set by these two targets, ranging from 20 ~ 50 % of the entire ocean area (De Santo, 2013). Growing awareness of the importance of MPAs is expected to promote the establishment and expansion of more protected zones worldwide.

Many studies have reported MPA success in depleted fish stocks (e.g. Polunin and Roberts, 1993; Mosquera et al., 2000; McClanahan and Arthur, 2001; Halpern, 2003; Aburto-Oropeza et al., 2011; Chirico et al., 2017; Fidler et al., 2017), and some in restoring coral and other habitat-forming organisms by limiting human activities (Pandolfi et al., 2003; Mumby et al., 2007; Diaz-Pulido et al., 2009; Mumby and Harbone, 2010). In the context of coral protection, coral within MPAs receive less anthropogenic impacts, specifically fishing, and hence are more likely to recover compared to outside MPAs and become resilient to disturbances such as climate change (McLeod et al., 2009). Furthermore, MPAs limit the coverage of microalgae by restoring the population of herbivorous fishes, which can in turn facilitate the coral recruitment (Hughes et al., 2007; Mumby et al., 2007). Protected coral reefs, however, still remain vulnerable to risks that arise from beyond MPAs (Mora et al., 2006). In other words, MPAs alone are not necessarily beneficial to coral when other factors, both local and large-scale, outweigh the protection effect (Jones et al., 2004; Cox et al., 2017), but can be effective when used in conjunction with other management measures (Allison et al., 1998).

1.2.3 Long-term monitoring: values and difficulties in habitat protection

Long-term monitoring is an imperative aspect of habitat management to evaluate the effect of protection, inform stakeholders, and adjust governance of protected areas (Lubchenco et al., 2003; Pomeroy et al., 2004; Toropova et al., 2010; Addison, 2011; Jone et al., 2011). Protection does not necessarily result in the recovery of habitats (see Jones et al., (2004) and Cox et al., (2017)). Knowing how the status of habitats, including coral reefs, has changed within the boundaries of protected zones is essential to ensure that conservation objectives (including but not limited to recovery of habitats) are met. Tracking long-term habitat change can help detect and identify issues slowing down habitats to recover, help solve, or mitigate such issues, and eventually enhance the protection effect.

While many studies exist on long-term monitoring of MPAs using fish abundance as a metric of the protection effect (e.g. Abesamis and Russ, 2005; Alcala et al., 2005; Barrett et al., 2007; McClanahan et al., 2009), much fewer have measured the long-term impacts of MPAs on habitats. One study found that the benefits of MPAs to coral appear to increase with the number of protection years, and that increases in coral cover can be perceived about 15 years after the establishment of MPAs (Selig and Bruno, 2010). This supports the notion that long-term monitoring should be implemented and sustained to confirm the effectiveness of MPAs (Addison, 2011).

Understanding the interactions between MPAs and habitats over multi-year time periods is challenging. Benthic communities can be affected by various factors, both local and global-scale, ranging from destructive fishing to ocean pollution to climate change (Mumby et al., 2007; McClanahan et al., 2008; Côté and Darling, 2010; Olds et al., 2012). Furthermore, the extent to which these attributes influence protected habitats can vary by years or locations. Physical and social attributes of MPAs (e.g. MPA area or enforcement level) can also affect habitat management and thus, protection effects (Magdaong et al., 2014). These factors are often taken into account when modeling MPA effects. However, variable selection can be arbitrary, and it is difficult to measure all possible influencing factors (Yasué et al., 2010; Molloy et al., 2013). Additionally, some factors are difficult to quantify, particularly measures of enforcement and compliance with MPA rules (Coglianese, 2012). Any of these factors can make the assessment of MPA long-term protection highly complicated.

Limitations of sampling methods add to difficulties in long-term monitoring of MPAs. Selection of sampling sites and areas, sampling time and frequency, and sample sizes often get biased by the researcher's perception and hence restrict the efficiency of sampling techniques, especially underwater (Leujak and Ormand, 2007; Yasué et al., 2010). Reasons include the fact that time available to collect data is often limited by the amount of air in SCUBA tanks and highly variable ocean conditions (Leujak and Ormand, 2007). In addition, the cost of surveys and the community support for MPAs can confine the capability of methods to comprehensively measure the habitat protection effect (Baldwin et al., 2005; Jokiel et al., 2015).

In the ocean, as on land, population assessments often depend on line transects, in which distributions of organisms are recorded along a transect line (Bowker et al., 2012; Facon et al., 2016). They also commonly use quadrats, in which individuals or species are counted within a quadrat frame (McCarthy et al., 2013; Biggerstaff et al., 2017). In the quadrat method, a delineated space may be photographed and later digitized. A third approach is to use plotless sampling in which distances between randomly plotted points and the closest individual are measured (Manly and Alberto., 2014). Such plotless sampling has most often been used for surveys of forest vegetation (Zhu and Zhang, 2009), but has also been suggested for coral reef surveys (Loya, 1978).

Any long-term assessment of environmental trends is especially difficult when sampling methods change over the course of monitoring. In theory, of course, an effective monitoring program should not change sampling methods over time (Conway and Gibbs, 2001). That said, changes in standard methods are sometimes necessary and important. One reason might be a failure in a first sampling method to yield appropriate data (e.g. difficulties in maintaining the fixed transects). Other possible reasons are when more cost-effective sampling emerges or is required to collect data due to limited budgets, or when the current method is logistically difficult to conduct for some reasons. Changing the method might cause differences in various aspects such as sample units, sampling areas, and sample sizes. Such alterations can affect results when one tries to combine data from different methodologies.

Where more than one method has been used, careful evaluation and integration is required to understand how, together, they can estimate change across sampling periods. To date, however, little effort has been invested on standardization of data from different methodologies. One study of terrestrial fields comprehensively discussed how to cater to inconsistency of data in case when multiple methods were involved in environmental monitoring (Beard et al., 1999). One suggestion was to look at correlations between old and new methods then use conversion factor to convert between the two sets of the results. Another suggestion was to introduce a variable into the analysis to estimate and account for differences in methods.

1.3 Research objectives and thesis outline

My research is framed within a goal of exploring ways to synthesize data from different sampling methods using data from long-term monitoring on MPAs in the central Philippines. This survey has been conducted since 1998 and involved two commonly used sampling techniques; line intercept transect (LIT) and photoquadrat (PQ). LIT was conducted in the first half of monitoring (1998 ~ 2010) and PQ in the later half (2007 ~ 2016), but these two surveys were conducted simultaneously between 2008 and 2010. Data from the method transition period allowed us to assess the compatibility of these two sampling techniques and verify the validity of treating a whole set of historic data as one time-series. Thus, my research was directed at answering the following two questions;

1. Are estimates of mean percent coral cover from different sampling methods comparable?
2. Do these methods allow detection of temporal patterns of percent coral cover in the same way (significantly increase or decrease, or no significant change over the overlapping period)?

This study provides a rare assessment of different sampling methods to understand increases or decreases in coral cover over multiple years rather than a one-year snap shot of estimates. My thesis concludes with a general discussion on my findings in this study, and how they contribute to further analyses of historical benthic data. Facing several data-related issues, including missing information and inconsistent data labeling, this study also discusses how data should be managed and suggests how long-term monitoring should be conducted for more effective MPA assessment.

1.4 Context and collaborations

This research took place in MPAs located in the Danajon Bank area of the Philippines, all of which are no-take marine reserves, some with associated buffer zones for local use. Danajon Bank is located off western and northern Bohol Island in the Visayas region, and is an area of international significance because of its double barrier coral reef formation and high marine biodiversity (Christie et al., 2006). The reef stretches across approximately 145 km, with an aggregated coastline of 699 km including 40 islands (Christie et al., 2006; Hansen et al., 2011; Yasué et al., 2012). Many communities in Danajon Bank rely heavily on marine resources for their food and livelihood (Samoilys et al., 2007). As a result, this area has been subject to high fishing pressure through not only destructive fishing methods such as dynamite and cyanide fishing, but also small-scale fisheries including hook and line, nets, and traps (Green et al., 2002; Christie et al., 2006; Selgrath, 2017). These fishing practices have devastated the condition of not only the marine resources, but also the coral reefs in Danajon Bank (Christie et al., 2006).

As of 2014, the Philippines had over 1,800 MPAs (Cabral et al., 2014). About 30 had been nationally designated through the National Integrated Protected Areas System Act of 1992 while the rest were community-based and created primarily for fisheries management purposes (The Coral Triangle Atlas, 2017). In Danajon Bank, approximately 100 MPAs have been established (Hansen et al., 2011). Of this number, 35 MPAs were generated as no-take marine reserves by local communities with the assistance of Project Seahorse Foundation for Marine Conservation (PSF).

PSF is a Filipino non-profit organization for marine conservation that has worked in Danajon

Bank since 1995 and is a counterpart of Project Seahorse at the University of British Columbia (UBC). PSF has continued contributing to marine conservation in this region as part of Zoological Society of London (ZSL) since 2015, and remains active in establishing and supporting MPAs with local communities. PSF / ZSL - Philippines has conducted long-term monitoring programs on PSF / ZSL-assisted MPAs since 1998. This long-term monitoring is aimed at assessing the effectiveness of MPAs in terms of fish abundance and diversity, seahorse populations, and benthic communities, especially hard coral and seagrasses. Data analyzed in this study were all collected by the biologists at PSF / ZSL - Philippines and constitute a core part of this research. They also kindly hosted me during their long-term MPA monitoring survey in the spring of 2016 to collect a new set of data and offered their generous support to sort out a vast amount of historic benthic data. The collaborative work with PSF / ZSL - Philippines has been critical to the success of my research.

Chapter 2: Comparison of line intercept transect and photoquadrat methods

2.1 Introduction

Only a limited amount is known about how marine protected areas (MPAs) affect habitats, with most research focusing on changes in fish populations. Theoretically, MPAs allow impacted habitats, including coral reefs, to recover by mitigating or excluding human activities within the boundaries of protected zones (Koenig et al., 2000). A 2.5-year investigation of ten sites inside and outside a fully protected Bahamian marine reserve demonstrated that increases in coral cover were higher at reserve sites than those at non-reserve sites (Mumby and Harbone, 2010). Furthermore, a meta-analysis of annual changes in coral cover in 310 MPAs revealed that the benefits of MPAs in terms of coral recovery appear to increase with the number of protection years (Selig and Bruno, 2010). MPAs are, however, not the panacea for habitat recovery. Some studies of individual reefs or small groups of reefs found that MPAs did not prevent coral loss and other forms of reef degradation. For example, one study documented a decline in coral cover from 66 % in 1996 to 7 % in 2002 through its annual surveys of four marine reserves in Papua New Guinea (Jones et al., 2004). This study attributed the decline to a combination of coral bleaching, gradual increases in sedimentation from terrestrial run-off, and outbreaks of crown-of-thorns starfish rather than mismanagement of the marine reserves.

Given uncertainties in MPA impacts on habitat recovery, efforts to evaluate the response of habitats to protection will be important for enhancing MPA performance. This is true whether we are focusing on habitat-forming organisms themselves or the fishes and invertebrates that live among them; the quality of habitats often influences biomass and abundance of marine life, and hence of marine resources. Habitat degradation is likely to reduce the ability of fish stocks to replenish themselves (Christie et al., 2006). At the same time, the high productivity of habitats, specifically coral reefs, implies high fisheries yields (Roberts et al., 2002). A 12-year study in Mombasa Marine National Park in Kenya detected an increase in catches outside the reserve with an increase in coral cover inside the reserve (Rodwell et al., 2003). In addition, an 8-year study of marine reserves in Papua New Guinea revealed that the greater the dependence of fish species on living coral, the greater the observed decline in fish abundance when coral decreased (Jones et al., 2004).

Although several techniques are commonly used to evaluate habitat protection, it remains difficult to determine how MPAs affect habitats. This is primarily because many complex, interacting processes (e.g. impacts of climate change, pollution, and physical destruction often associated with destructive fisheries) affect benthic communities, particularly coral reefs (Mumby et al., 2007; McClanahan, 2008; Côté and Darling, 2010; Olds et al., 2012). Biological surveys can also be influenced by researcher's perception in dimensions ranging from sampling schemes to selections of variables for models (Yasué et al., 2010). Given these conditions and that surveys sample only a subset of populations, all sampling techniques suffer from the issue of accuracy and precision (Andrew and Mapstone, 1987). Accuracy is referred to as the ability to estimate the true values while precision is the ability to detect small changes (Leujak and Ormand, 2007). Therefore, precision should be considered as important along with accuracy when comparisons of successive estimates, such as long-term monitoring, are the primary goals (Andrew and Mapstone, 1987).

Commonly used techniques in MPA assessment include manta tow, line intercept transect (LIT), line point transect (LPT), photoquadrat (PQ), and video transect. Each technique has its own tradeoffs which impacts data type, resolution, and quality (English et al., 1997; Nadon and Stirling, 2006; Leujak and Ormand, 2007). Manta tows involve towing an observer behind a boat with a rope to which a manta board is attached (English et al., 1997). This technique is appropriate to assess broad changes in benthic communities in relatively large areas. However, since the cover of sessile lifeforms is visually estimated while an observer is being towed, they could easily be overlooked, especially in a condition of the poor visibility. Therefore, estimates could lack accuracy.

Other sampling techniques mentioned above deploy observers who record data using measuring tapes or other recording tools. These methods are able to generate detailed and accurate data on a fine scale. In the LIT survey, surveyors identify lifeforms encountered under the transect using benthic categories and measure the length of each of them (English et al., 1997). On the other hand, the LPT survey implements counting the number of times lifeforms were found under randomly or systematically placed points along the transect (Nadon and Stirling, 2006). Since neither LIT nor LPT requires expensive equipment other than SCUBA gear, both surveys are relatively cheap to conduct. However, they require longer survey time and more sampling effort compared to the other methods that employ recording tools (Leujak and Ormand, 2007). In the PQ survey, digital images of substrates are taken along the transect whereas in the video transect survey, benthic communities are filmed as movies. Using recoding tools to collect data allows surveyors to gain sufficient amount of data with less sampling effort and save a fair amount of time in the field (Leujack and Ormand, 2007). However, it could be offset by lengthy data digitization (quantification of benthic cover) and analyses required in the laboratory (Leonard and Clarke, 1993; Molloy et al., 2013).

Despite the widespread use of these sampling methods, little effort has been invested in understanding how each might influence our understanding of MPA performance. Given different advantages and disadvantages of sampling methods, interpreting technical approaches is vital to better understanding MPA performance. One study comprehensively compared six coral community survey methods (Leujak and Ormand, 2007). It reported that line-based techniques such as LIT and LPT tended to overestimate hard coral cover compared to quadrat-based techniques such as PQ. They primarily attributed this observation to the contour effect. The transect line is likely to follow the surface of coral colonies, tending to estimate the circumferential area of coral, which could appear to be larger than planar areas. A more recent study also reported that LPT generated higher coral cover estimates compared to eight other methods, including PQ, although differences among these methods were not significant (Jokiel et al., 2015). They also attributed this finding to the contour effect which resulted from slack transect lines. However, these two studies contrasted with each other in that the former recommended using quadrat-based techniques, including PQ, as preferred sampling methods over line-based techniques whereas the latter noted that different sampling methods produced similar estimates and hence were all comparable with each other.

Here, I had coral cover data collected from 1998 through 2016 for eight MPAs in the Philippines. This long-term monitoring program involved a sequential use of two common techniques – LIT and PQ, with some temporal overlap between them. The LIT survey was conducted from 1998 through 2010, and the PQ survey from 2007 through 2016. Since data were produced from two different methodologies, the precision of estimates might be deteriorated (Leujak and Ormond, 2007). In addition, each survey had its own unique challenges. The LIT survey initially used supposedly fixed transects, but it turned out that they did not remain in a fixed position. However, samples did not include enough transects to allow them to be treated as statistically haphazard. As for the PQ survey, transects were supposed to be laid randomly with greater sample sizes. However, we were unable to guarantee that observations were independent due to a decision to place the transects closely to each other, simplifying sampling, which could have led to spatial autocorrelation.

Small sample sizes and spatial autocorrelation are common problems in both terrestrial and underwater surveys (e.g. Bremer and Farley, 2010; Ateweberhan et al., 2011; Hackerott et al., 2013; Meyers et al., 2014). However, problems with small sample sizes commonly refer to the number or individuals observed rather than to unreliably fixed transects as in my study (e.g. Collard et al., 2009;

Donham et al., 2017). In addition, I could not find any study both on land and in the sea which report transects deployed closely to each other. Spatial autocorrelation can be incorporated into models in data analyses (Zuur et al., 2009b), but I worried that our data might have been dependent to an extent that this issue could not be solved by adopting model structure.

Having faced these challenges, I initiated this study with one major question; how can we synthesize data from two methodologies? Fortunately, an overlapping period of the LIT and PQ surveys (2008 ~ 2010) allowed us to compare data from two sampling techniques. The comparison of different methods in terms of coral cover has been investigated by several researchers (e.g. Brown et al., 2004; Lam et al., 2006; Nadon and Stirling, 2006; Leujak and Ormand, 2007; Jokiel et al., 2015). However, there are relatively few studies investigating how different methods estimate temporal patterns of coral cover. This study, therefore, analyzed data from the method transition period to investigate: (1) how mean percent coral cover estimates during these years differed depending on whether the LIT or PQ surveys were used and (2) whether LIT and PQ estimated temporal patterns of coral cover in the same ways. Before addressing these topics, I attempted to increase the sample size of LIT data (subsampling LIT data). I then assessed the benefit of increasing the sample size. Additionally, I explored possible spatial autocorrelation of PQ data. Finally, I evaluated the costs and benefits of altering sampling methods in MPA long-term monitoring.

2.2 Materials and methods

2.2.1 Study area and long-term monitoring

2.2.1.1 Study site

Danajon Bank in the central Philippines is known for its unique double barrier reef formation, consisting of inner and outer reefs, a suite of inshore islands, patch and fringing reefs, seagrass beds, and mangroves (Christie et al., 2006; Hansen et al., 2011). It is a 145 km long coral reef system stretching along the western and northern coast of Bohol (Yasué et al., 2012). Its reef comprises an area of 272 km^2 and makes up over 1 % of the total area of coral reefs in the country (Christie et al., 2006). Local communities depend heavily on marine resources, but overexploitation and mismanagement have led to depleted fisheries and damaged marine habitats (Green et al., 2002; Christie et al., 2006). The many and diverse fisheries activities in this region include use of illegal methods such as trawling, blast, and cyanide fishing, and numerous small-scale fisheries, which have largely devastated coral reefs in Danajon Bank for the past few decades (Green et al., 2002; Christie et al., 2006; Selgrath, 2017). In addition, the condition of reefs has been degraded by sediment accumulation, which impedes the growth of hard coral (Christie et al., 2006).

I here report on studies undertaken on community-managed MPAs catalyzed by Project Seahorse Foundation (PSF: now integrated into Zoological Society of London - Philippines or ZSL - Philippines), a Filipino non-profit organization for marine conservation that has worked on Danajon Bank since 1995. The first MPA of this kind (Handumon) was established in 1995, with community engagement helping to prompt establishment of a total of 35 MPAs as no-take marine reserves. I report on the eight MPAs where long-term monitoring programs have been executed twice a year on regular basis. These eight MPAs were established between 1995 and 2004 and thus, ranged from 12 ~ 21 years in protection duration as of the sampling year in 2016 (Table 2.1). They were also different in size (surface area in ha), compactness (defined as $4*\pi*A / P^2$ where A is the area, and P is the perimeter of the MPA: Li et al., 2013), distance from the nearest village, and ecological zone with two levels of locations: (1) coastal (between the shore and inner reefs) and (2) reef (between the inner and outer reefs). In addition, percent coral cover in the year of establishment differed among MPAs. Due to different years of establishment, the first survey years differed among MPAs (Table 2.2).

2.2.1.2 Long-term monitoring on MPAs

Long-term monitoring, which began in 1998, was primarily directed at measuring the effectiveness of protection in terms of fish abundance and diversity, but also to track the seahorse populations and communities of benthic habitats (especially hard coral and seagrasses) (Suarez, 2009). The project team consisted of the biologists in PSF, local fishers, and volunteers, and conducted a series of surveys including a visual census of fish species and abundance, and the quantitative benthic assessment (McCorry, 2005). Data were collected inside and outside MPAs, and at reference sites, which were located at least 1 km from the boundaries of the nearest MPA. The reference sites were intended to reflect baseline changes, without immediate influence from MPAs. They are important because MPAs should be influencing areas immediately near them in a way that might obscure possible changes between protected and non-protected zones. A series of surveys was conducted with the assistance of community organisers who facilitated relations with local communities in Danajon Bank.

Surveys were conducted twice a year (dry season: March ~ April, wet season: September ~ October) from the wet season in 1998 (herein Wet 1998) through the dry season in 2016 (herein Dry 2016). Long-term monitoring on benthic habitats involved a sequential use of two sampling methods: (1) line intercept transect (LIT) and (2) photoquadrat (PQ). Monitoring was initially done using the LIT survey, then supplemented by the PQ method from Dry 2007, and undertaken using only the PQ survey from Dry 2011. Both methods were conducted between Wet 2008 and Wet 2010 to investigate comparability of data from two different methodologies (Table 2.2). In spring 2016, I joined the monitoring survey team led by the biologists at ZSL - Philippines to collect a new set of data from 18 MPAs, including eight MPAs observed in this study, and at five reference sites.

2.2.2 Sampling methods

2.2.2.1 Line intercept transect method (LIT)

2.2.2.1.1 LIT sampling protocol

The LIT survey was conducted from Wet 1998 through Wet 2010. Eight 50 m fixed transects were set up parallel to the shore for each MPA, with four located inside and other four outside MPAs (Fig. 2.2). Furthermore, two of four transects inside and outside MPAs were set up on the shallow reef crest (< 5 m: herein shallow transects) and other two on the deep reef slope (5 ~ 10 m: herein deep transects) (Fig. 2.3). Similarly, four transects were laid for each reference site, out of which two were located on the shallow reef crest and the other two on the deep reef slope.

In this survey, one diver laid a measuring tape along the first 20 m of the transect, and another diver recorded the position where substrates changed to 1 cm resolution along the transect (i.e. an observer measured a range of the measuring tape crossed by each of different substrates) (Suarez, 2009). The substrates were categorized into eight major groups by benthic habitat type such as coral, algae, and seagrass. Each major group was subdivided into sub-categories by life or structural form (e.g. branching coral, massive coral etc. for coral) to create a total of 31 sub-categories for eight groups, and the standardized code was used to record each sub-category (Table 2.3: modified from English et al., 1997). Descriptions of each benthic sub-category are provided in Appendix A.

2.2.2.1.2 Issues of the LIT survey

Over time, the LIT sampling began generating concerns about reliability and utility, primarily because of the mobility of the supposedly fixed LIT transects: (1) transects shifted from the deep to shallower water by typhoons or fishing gears; (2) transects broke or were lost; and (3) it was difficult to locate supposedly fixed transects. Resultant haphazard locations of transects meant that they had to be regarded as mobile rather than fixed, with a consequent loss of power due to small sample sizes (sample n = 2 for shallow and deep areas each: inside and outside the MPA taken as two different sites). A lack of power could inflate likelihood of the Type II error when coral cover was estimated.

In 2007, we added a new sampling method which allowed the use of haphazard rather than fixed transects with larger sample sizes, based around PQ. A PQ pilot study was conducted from Dry 2007 ~ Wet 2008 (although the survey in Dry 2007 did not yield good quality images for technical issues such as malfunction of digital cameras). Using data from two MPAs (Batasan and Handumon) and two reference sites (Canlangi and Putik) collected in the pilot study, our colleagues investigated the minimum number of transects powerful enough to detect relative changes in coral cover (with a power of 80 % and at $\alpha = 0.05$: Molloy et al., 2013). Based on their analysis, the optimal sample size was estimated at 10 transects with 23 quadrats nested within each transect and 10 points per quadrat.

2.2.2.2 Photoquadrat method (PQ)

2.2.2.2.1 PQ sampling protocol

The PQ survey began in Dry 2007 and has been continued ever since, overlapping with the LIT sampling for three years before the latter was phased out (with no LIT survey conducted from Dry 2007 ~ Dry 2008). Photoquadrat sampling aimed to characterise benthic habitat composition and percent cover of sessile lifeforms using haphazardly laid transects with increased sample sizes. Twenty transects were placed using transect tapes for each MPA, out of which 10 were located inside and other 10 outside MPAs. Ten transects were also placed in each reference site. Each transect was 23 m long and placed at the depth of 3 ~ 4 m on the reef crest, which corresponded with the shallow depth in the LIT method, along constant depth contours.

In this survey, a diver photographed benthic habitats along the transect at 1 m intervals, for a total of 23 quadrats on each transect. Each digital image was taken 60 cm above the substratum and captured an area of approximately 0.25 m^2 (0.5*0.5 m). A monopod was used to maintain the same distance between the camera and substratum. Another diver measured the rugosity of the sea bottom using the brass chain along the first 10 m of the same transect simultaneously. To digitize images in percent cover data, digital images were imported into Coral Point Count with Excel extensions software (CPCe) in the laboratory (Kohler and Gill, 2006). Ten points were randomly spotted over each image, and benthic types under each point were identified and categorized into nine major groups and 35 sub-categories (Table 2.3: one additional major group and four additional sub-categories).

2.2.2.2.2 Issues of the PQ survey

Logistical constraints in the PQ survey mean that all transect tapes were laid out end to end in one long line within each sampling area; tapes were first placed with ends closely to each other, but then the field biologists began to connect them end-to-end during the PQ survey period although without documenting these changes (Fig. 2.4). The biggest limitation was the small area of most MPAs, and of their coral reefs, which just did not allow for much spacing between transects if replication was important. Other logistical constraints included transect tapes that could be easily lost when laid separately and the separation of transect tapes requiring more survey time while the amount of air in SCUBA tanks are limited. To separate transects, transect tapes with extra lengths (25 ~ 26 m) were used to make 2 ~ 3 m intervals between them (Fig. 2.5). However, since the deployment of transects did not fully meet random observations, I needed to consider possible spatial autocorrelation.

2.2.3 Data preparation

2.2.3.1 Cleaning data

Owing to several data errors (Appendix B), the datasets were cleaned to make them into an appropriate format for the analysis. LIT original data (raw filed data) were saved as Excel spreadsheet files containing every fraction of the transect (rather than summed lengths of each benthic category within the transect); each file contained data of multiple sites collected from all years of the surveys. PQ original data (data before cleaning) consisted of (1) digital images of substrates, (2) CPCe files (digital images with 10 random points), and (3) Excel spreadsheet files digitized by CPCe from benthic images, all of which were saved by quadrat or transect for each site and year.

First, I obtained historic benthic data files collected in the Philippines from the database of Project Seahorse at UBC. Second, I extracted LIT data and made new spreadsheets by transect for each site (each spreadsheet containing data from all years). At the same time, I sorted out all types of PQ data files by year, season, and transect for each site. I then compiled all missing data files as data tables for both LIT and PQ data (Appendix C). Third, I obtained additional data files from the long-term monitoring database at ZSL - Philippines, filling in gaps in data held by Project Seahorse. Where temporal gaps in data collection still remained, I queried our Filipino colleagues. The main reasons for gaps mentioned by the biologists were limited budgets, limited time, limited human resources, and broken transects (only for LIT). This investigative process was intended to distinguish areas with zero cover from areas with no data in the database. Lastly, after sorting out data files, I changed spreadsheet data into an appropriate format as csv files for the analysis of data from the two methods (Appendix D; I report steps taken to transform data with examples of file names and locations of data files for each step). In the format transformation, I arranged data within each file chronologically and numerically for transect (LIT) and quadrat (PQ) information according to the number assigned to them. I also corrected mislabeling in data (i.e. checking unlikely benthic codes, or codes assigned to information about each transect or quadrat). Each transect and quadrat was coded by the biologists in a systematic way with MPA, location, year, season, transect, and quadrat codes (Appendix E).

2.2.3.2 Subsampling Treatments

2.2.3.2.1 Issues of data to fix by subsampling

Because of the shift in the sampling design, I encountered several data analysis issues: (1) the supposedly fixed LIT transects were actually somewhat haphazard in their locations; (2) sample sizes of LIT data became very small once they were recognised as haphazard; and (3) PQ data might exhibit spatial autocorrelation because transect tapes were laid in one long line. Lastly, I faced the challenge of combining data from LIT and PQ (a main focus in this study). To explore how subsampling already collected data might help address these three limitations, I explored the five subsampling treatments of the LIT and PQ data, described below.

2.2.3.2.2 Subsampling LIT data

To solve the issue of statistical power resulting from unreliably fixed transects and low sample sizes in LIT, I assumed all transects to be haphazard. I then subsampled the transect data to create ‘quadrats’ to serve as the sample unit based on cleaned data on the transformed spreadsheet files. Using R statistical software, an imaginary 0.5*0.5 m quadrat (herein LIT quadrat) was projected at 1 m intervals along the 20-m fixed transect data, and 10 random points were superimposed along the linear transect data within each newly created LIT quadrat. These points were then matched with a corresponding benthic category (Table 2.3), summed up by category, and divided by 10 (the number of random points projected for each quadrat) to obtain percent values for each category within each quadrat. This method produced 20 samples per transect and thus, 40 samples for a shallow site and 40 for a deep site (total 80) per sampling survey in one site. Here, I call LIT data before subsampling "**LIT line data**" and after subsampling "**LIT point data**" (Table 2.4).

One remaining risk is potentially some non-independence of LIT data. Although the transects in the LIT survey could not be treated as fixed, surveyors did attempt to set measuring tapes where the fixed transects were suspected to have been. The transects could thus have overlapped with their locations in previous years such that observations might not have been fully independent within each site. This risk has to be carefully assessed by cross-checking results using multiple tests available. Comparison of results help us understand to what extent non-independence of LIT data has affected analyses and thus, conclusions drawn from such data.

2.2.3.2.3 Subsampling PQ data

To address issues with the PQ transects, I treated 10 connected transects as one line and took 23 quadrats per transect as the sample unit. It produced 230 samples each for inside the MPA, outside the MPA, and the reference site, respectively. Having a quadrat as the sample unit also helped increase the sample sizes of PQ data and standardize the data unit when combined with LIT data. Furthermore, 80 or 40 quadrats were randomly selected from each line to subsample PQ data using R. This was to create spatial lags between quadrats and explore the optimal sample sizes of PQ data when data from the two methods were combined. These sample sizes were chosen because the sample sizes of subsampled LIT data were 80 per site when data from both shallow or deep transects were used or 40 per site when data from shallow transects were used. Here, I referred to PQ data with all 230 quadrats as "**PQ 230 data**", and PQ data with 80 or 40 quadrats as "**PQ 80** or **PQ 40 data**" each (Table 2.4).

2.2.3.3 Creating subsets of data for the analysis

I created subsets of data prior to the analysis, drawing only on data that were collected using both LIT and PQ methods during the same sampling survey in the same MPA (Wet 2008 ~ Wet 2010). First, data for shallow transects were extracted from both LIT line and LIT point data so that I could compare methods (no deep data were collected in PQ). Second, data from inside MPAs were extracted. Outside MPA and reference site data were not used in this study because the focus here was to compare LIT and PQ data rather than to investigate the protection effect on coral. Asinan MPA consisted of two distinct areas (Guard House and Post areas), of which I used on the Guard House area. Third, information of seven sub-categories under the major category of coral were extracted (Table 2.3) and summed to obtain values of percent coral cover per transect or quadrat.

2.2.4 Data Analysis

2.2.4.1 Comparisons of LIT and PQ data in mean percent coral cover

I conducted two sets of paired analyses: primary paired analyses comparing mean percent coral cover for all MPAs combined (main analyses) followed by secondary paired analyses with the nested structure to account for the possible non-independence of LIT data. When the LIT and PQ surveys were conducted in the same site, season, and year, these were considered paired samples. Since I was interested in exploring subsampling treatments that would make the two field survey methods more equivalent, I assessed six combinations of the treatments for subsampling the original data: (1) LIT line and PQ 230 data; (2) LIT line and PQ 80 data; (3) LIT line and PQ 40 data; (4) LIT point and PQ 230 data; (5) LIT point and PQ 80 data; and (6) LIT point and PQ 40 data (Table 2.4).

First, I calculated mean percent coral cover by sampling survey for the five treatments tested here. Surveys were conducted two or three times at each MPA between Wet 2008 and Wet 2010, adding up to 22 sampling surveys across eight MPAs and hence, generating 22 mean values per subsampling treatment (survey n = 22). Each pair was tested based on measures of mean percent coral cover taken during the overlapping period of the two methods. Within each survey, the measures of mean percent coral cover are based on a different number of samples depending on the treatment (i.e. sample n = 2 for LIT line data, sample n = 40 for LIT point data etc.), but each analysis evaluated differences in coral cover between treatments of 22 mean values.

Second, I inspected distributions and variance of data statistically and visually to determine the appropriate type of tests. The Shapiro-Wilk test determined that values of all subsampling treatments were normally distributed (Table F.1). However, the histograms showed skewed distributions of data points and thus, I concluded that they did not fully follow the normal distribution (Fig. F.1). To investigate the variance of data, I visualized mean values as scatter plots, and the F-test was conducted to examine whether values of each treatment had equal variance in pairs. Variance of data was not significantly different between two treatments in all pairs (Fig. F.2 and Table F.2). However, since the assumptions of the parametric t-test were not fully met (skewed distributions of data), I determined to use the non-parametric Wilcoxon signed-rank test. In addition, since my interest lied in the difference between the two methods (LIT and PQ) rather than a difference among the five treatments, this test was performed instead of the Friedman test (a non-parametric test to detect differences among multiple groups, which is followed by the post-hoc test).

Third, I ran the Wilcoxon signed-rank test to compare LIT and PQ data for the six pairs. Since I carried out multiple comparisons in both the Wilcoxon test and a series of preliminary analyses, the possibility of the Type I error could inflate (Benjamini and Yekutieli, 2001). Therefore, a p-value of 0.05 was adjusted using the Bonferroni procedure by dividing this value by the number of tests performed (Armstrong, 2014). Data were analyzed using the wilcox.test function in R.

For the second set of analyses, I tested the same six pairs of subsampling treatments using nested structure. This set of analyses allowed me to evaluate whether data among different years within one MPA were independent, as assumed by the Wilcoxon test. The nested analysis used the lme function of the nlme package in R, and random effects of "site" and "sampling survey" nested within site were included in the model. Note that there is no non-parametric test which can include two random effects. The results produced from this test were found to be similar to the Wilcoxon test (Appendix G).

2.2.4.2 Temporal patterns of percent coral cover

I estimated temporal patterns of percent coral cover over the three-year study period using the linear mixed effects and linear models. Subsampling treatments assessed in the analyses were: LIT line data, LIT point data, PQ 230 data, PQ 40 data, LIT line and PQ 230 combined data (herein LIT line + PQ 230 data), and LIT point and PQ 40 combined data (herein LIT point + PQ 40 data) (Table 2.4). I then compared slopes produced from each treatment to investigate differences in coral cover trends. All data points were used to estimate slopes, rather than taking mean values.

First, I used the linear mixed effects model to test temporal patterns of coral cover for all MPAs combined. Year was modeled as the explanatory variable and percent coral cover as the response variable. The term of year increased by 0.5 (i.e. 2008.5, 2009.0, 2009.5, etc.), reflecting two sampling surveys per year with 0.5 representing the wet season and without 0.5 the dry season (i.e. 2009.0: Dry 2009, 2009.5: Wet 2009). In addition, site (MPAs) was included as a random effect to account for the repeated observations. The lme function of the nlme package in R was used to run the model. The model assumptions were then tested to evaluate the model validity. The assumption of residuals having constant variances was tested by plotting residuals against fitted values and years. The assumptions of residuals normally distributed and having a mean of zero were examined using Q-Q plots (plots to compare the sample distribution function and a theoretical distribution function) and histograms. I also ran the generalized least square model using the gls function in the same package. The response and explanatory variables were the same as the linear mixed effects model, but this model did not include the random effect. I then compared the two models (with and without the random effect) using AIC to confirm that including the random effect improved the model.

The generalized least square model without any optional structure such as autocorrelation or heterogeneity produces the same results as the linear model (Zuur et al., 2009a). However, the linear model uses the OLS estimation implemented with the help of the QR decomposition of the design matrix (Gałecki and Burzykowski, 2013). In contrast, a default estimation method of the generalized least square model is the restricted maximum likelihood (REML: the estimation method used here). REML is preferred over the maximum likelihood (ML) when models with different variance structures are compared because REML is less biased in its calculation of variance components than ML (Pinheiro and Bates, 2000). REML was used for the linear mixed effects model here as well.

Second, I fitted the linear regression model to test the trends of coral cover over years for each MPA. In this analysis, year was modeled as the explanatory variable and percent coral cover as the response variable. The term of year increased by 0.5 to differentiate the two seasons. The lm function in R was used to conduct the linear model. Each model was then examined as to whether it met the model assumptions as described above for the linear mixed effects model. When results of all subsampling treatments were compared (both linear and linear mixed effects models), a Bonferroni corrected critical value was used to determine the significance of slopes for all analyses.

2.2.4.3 Spatial autocorrelation of PQ data

The assumption of spatially independent observations was possibly violated by the PQ survey given that transect tapes were laid end to end. Therefore, I explored spatial autocorrelation for PQ 230 data collected in the wet and dry seasons in 2010. Semivariograms were plotted for each of the two seasons by MPA using R. Data from these sampling surveys were selected because all eight MPAs were surveyed using the PQ method. When spatial lags of quadrats were set, I assumed that intervals between transects were always 2 m; some intervals were apparently 3 m, but such haphazard differences were not recorded. For Asinan, only one quadrat was available for even numbered transects because 22 other quadrats were lost (possibly because of data entry errors). Thus, spatial lags were set to reflect 22 missing quadrats every other transect for this MPA.

2.3 Results

2.3.1 Comparisons of LIT and PQ data in mean percent coral cover

LIT data (both LIT line and LIT point data) generally indicated higher mean percent coral cover than PQ subsampling treatments (PQ 230, PQ 80, and PQ 40 data). However, LIT line data showed a significant difference when compared with PQ data whereas LIT point data did not. Coral cover estimates by LIT line data were significantly higher than PQ 230 and PQ 80 data ($p = 0.007$ and 0.002, respectively: Figs. 2.6a and 2.6b, and Table H.1). The coral cover estimated by LIT line data was higher than PQ 40 data, but this difference was not significant under the Bonferroni corrected p-value of 0.0083 ($p = 0.025$: Fig. 2.6c and Table H.1). When LIT point data were compared with PQ 230, PQ 80, and PQ 40 data, the estimates of coral cover were not significantly different between the two methods ($p = 0.043$, 0.046, and 0.079, respectively: Figs. 2.6d, 2.6e, and 2.6f, and Table H.1).

2.3.2 Temporal patterns of percent coral cover

In our estimate of the temporal patterns of percent coral cover for all MPAs combined, PQ data or the combination of LIT and PQ data indicated that coral cover had decreased significantly (Fig. 2.7a and Table H.2.1). Subsampling treatments which indicated significant trends were PQ 230, PQ 40, LIT line + PQ 230, and LIT point + PQ 40 data, with estimates of decreases in percent cover ranging from -8 to -3 % per year ($p < 0.001$, except LIT point + PQ 40 $p = 0.003$). On the other hand, LIT data alone (LIT line and LIT point data) detected no significant pattern of coral cover over years.

When looking at each individual MPA (Figs. 2.7 b - i and Tables H.2.2 - 2.9), PQ data (PQ 230 or PQ 40 data) and the combination of LIT and PQ data (LIT line + PQ 230 data only) detected significant declines in coral cover in four MPAs (Bilang Bilangan, Batasan, Jandayan Norte, and Bantiguian). In contrast, LIT data (both LIT line and LIT point data) did not detect significant patterns of coral cover at all MPAs.

For Bilang Bilangan and Batasan, significant coral cover change was detected in three subsampling treatments, all of which derived from PQ data (PQ 230, PQ 40, and LIT line + PQ 230 data) ($p < 0.001$) (Figs. 2.7b and 2.7c, and Tables H.2.2 and G.2.3). The estimates for these MPAs by the three treatments were -18 to -9. No significant trend was found in other treatments, but both LIT line and LIT point data showed positive slopes, contrasting with other results.

For Jandayan Norte and Bantiguian, significant trends were detected by two subsampling treatments, both of which drew on PQ 230 data (PQ 230 and LIT line + PQ 230 data) ($p < 0.001$) (Figs. 2.7f and 2.7i, and Tables H.2.6 and G.2.9). The estimates for these MPAs by the two treatments were around -5 and -19. PQ 40 data also produced negative slopes, but they were insignificant for both MPAs. Additionally, the slope estimated by PQ 40 data for Jandayan Norte was steeper than by PQ 230 data; yet, the result from PQ 40 data was not significant. The slopes estimated by LIT line and LIT point data were positive except the estimate by LIT point data for Bantiguian, but neither treatment detected significant trends.

Temporal patterns of coral cover estimated for four MPAs (Pandanon, Asinan, Handumon, and Pinamgo) were mostly insignificant (Figs. 2.7d, 2.7e, 2.7g, and 2.7h, and Tables H.2.4, H.2.5, H.2.7, and H.2.8). Slopes estimated for Pandanon and Asinan were all insignificant except the estimates by PQ 40 data for Pandanon and LIT line + PQ 230 data for Asinan (Figs. 2.7d and 2.7e, and Tables H.2.4 and H.2.5). Slopes estimated for Asinan were relatively steep (> -15 except the estimate by PQ 40 data), but a significant trend was detected in only one subsampling treatment. No significant trend was detected in any treatment for Handumon and Pinamgo, indicating that coral cover remained constant over years in these two MPAs (Figs. 2.7g and 2.7h, and Tables H.2.7 and H.2.8).

2.3.3 Spatial autocorrelation of PQ 230 data

Spatial autocorrelation was not found in most MPAs (Figs. 2.8 and 2.9). When spatial autocorrelation was visualized as semivariograms, the elevation of semivariance values indicated the existence of spatial autocorrelation. However, our results did not exhibit clear increasing values in most semivariograms, indicating that PQ data were not spatially correlated. However, the extent to which semivariance fluctuated across transects differed among MPAs and between seasons.

PQ 230 data collected in Dry 2010 presented values of semivariance relatively constant across transects at six MPAs (Bilang Bilangan, Pandanon, Asinan, Jandayan Norte, Handumon, and Bantiguian) (Figs. 2.8a, 2.8c, 2.8d, 2.8e 2.8f, and 2.8h). In contrast, semivariance tended to increase across transects in two other MPAs (Batasan and Pinamgo; Figs. 2.8b and 2.8g). PQ 230 data collected in Wet 2010 did not exhibit obvious trends in any MPA except Bantiguian, where values tended to increase (Figs. 2.9a ~ h). The fluctuations of semivariance in Batasan and Pinamgo were smaller in Wet 2010 than in Dry 2010. In Asinan, they were smaller in the dry season than in the wet season.

2.4 Discussion

This study demonstrated that it is feasible, if challenging, to combine data from two of the dominant sampling techniques for benthic cover. Difficulties in integrating data from LIT and PQ methods into a single, informative time series arose because of the discrepancy in the deployment of transects and sample sizes between these two methods, coupled with several other issues (e.g. failure to maintain the fixed transects for LIT, possible spatial autocorrelation for PQ). Consequently, integrating data derived from different sources led to a dataset that differed in two ways: (1) mean percent coral cover estimated by LIT was higher than by PQ and (2) temporal patterns of percent coral cover indicated as significant by PQ were insignificant by LIT. However, I identified measures that facilitated integration of data into a single, comparable dataset, including subsampling data to increase low sample sizes of LIT data. The findings here also emphasized that surveyors should adjust the sampling scheme carefully if they switch methods during long-term monitoring. Data from the new method should be not only statistically powerful, but also compatible with previous data.

Researchers should examine differences in coral cover through the lens of accuracy and precision between sampling techniques when combining data from different sources (Green and Smith, 1997; Jokiel et al., 2015). This might specifically be the case when the estimates by LIT and PQ exhibited different estimates in a particular set of data, where a sequential use of multiple methods was involved. When estimated values of benthic cover from multiple methods are similar to one another, it indicates that they are comparable in accuracy (Leujak and Ormand, 2007). When these methods provide similar chances of detecting relative changes between years (or sites), it can be interpreted that they are comparable in precision (Leujak and Ormand, 2007). Such examinations can help understand how each method is affecting the results when assessing the protection effect. Interpreting data is a critical step prior to obtaining reliable pictures of the habitat response to protection in order to better manage MPAs.

In my study, increasing sample sizes of LIT data led to estimates of mean percent coral cover from the two methods (LIT and PQ) becoming more comparable with each other. The estimates by LIT data should be more accurate after subsampling as the statistical theory indicates that larger sample sizes improves the accuracy of estimated mean values (Yacci, 2009). Thus, subsampling likely helped improve the integration of LIT and PQ data with more accuracy and comparability.

The comparison of LIT and PQ confirmed the results reported by previous studies that LIT tended to estimate higher coral cover than other sampling methods (Weinberg, 1981; Ohlhorst et al., 1988; Leujak and Ormand, 2007). I found that both LIT line and LIT point data estimated higher mean percent coral cover than PQ data although the scale of differences was influenced by the way PQ data were subsampled. One study assessed the accuracy and precision of six survey methods in South Sinai, Egypt, with the quadrat mapping method as a baseline to evaluate estimate quality (Leujak and Ormand, 2007). Assuming that the quadrat mapping is the most accurate method, it discussed that coral cover was most accurately estimated by the video transect and least accurately by line point transect (LPT) and LIT. It also reported that all photographic methods tended to indicate lower coral cover while LPT and LIT generally yielded higher estimates.

A second study, on coral reefs in Hong Kong, compared coral cover estimates from the video transect operated by a remote operated vehicle (ROV) and SCUBA divers, and LPT (Lam et al., 2006). Its results showed that LPT recorded significantly higher coral cover than the video transect. Similarly, a third (more recent) study compared multiple methods, including random quadrat (both area and random point-based), LPT, video transect, manta tow, and PQ in Hawaiian coral reefs (Jokiel et al., 2015). It found that LPT estimated highest percent coral cover, followed by manta tow, video transect, area-based random quadrats, and point-based random quadrats although the estimates of coral cover among these methods were not statistically different from each other. These studies supported my finding of LIT prone to produce higher coral cover estimates than the other method.

Differences in coral cover estimates between LIT or LPT and the other methods can be attributed to four inherent characteristics of line-based sampling techniques: (1) contour effect; (2) view angle which is often coupled with an observer's bias; (3) proportion of substrates sampled; and (4) time inefficiency (Lam et al., 2006; Leujak and Ormand, 2007; Jokiel et al., 2015). First, the measuring tapes tend to slacken. As a result, they follow the contours of coral colonies even when they are thought be stretched tight. Loose transect tapes can lead to higher estimates of coral cover than methods where colonies are observed from above as the planar area. Second, it is difficult for observers to read the measuring tapes vertically underwater and thus, angles from which they observe the benthic substrate can differ. This is especially so when the tapes are off the sea bottom. Since the positions of the substrate beneath the tapes can vary depending on the view angle, it leads observers to a temptation to record any coral which might be below the line. Third, substantial differences in sampling effort can cause different estimates between LIT or LPT and quadrat-based methods such as PQ. Methods that utilise quadrats record a swathe of the sea bottom whereas LIT and LPT only do so linearly. Even when the same number of transects are used in line-based or quadrat-based methods, the total surface area covered in the survey is considerably smaller in the former methods. Such differences most likely result in different estimates among these methods. Lastly, LIT and LPT require significantly more time spent in the field than PQ or the other methods using recording tools (e.g. digital camera) since substrates are quantified *in situ*. However, achieving enough sampling effort to ensure the accuracy of estimates is logistically difficult in LIT and LPT; limited amount of air in SCUBA tanks and variable ocean conditions restrict the number of transects to be surveyed. The drawbacks of line-based methods have led to recommendations of using quadrat-based sampling techniques, especially PQ (Leujak and Ormand, 2007; Van Rein et al., 2011).

Unplanned changes in sampling protocols, such as differences in the deployment of transects between methods, might also have been a source of different estimates by LIT and PQ. The original fixed transects used in the LIT survey were supposedly set up on areas with relatively high coral cover because the primary purpose of long-term monitoring was to observe changes in benthic communities, specifically hard coral. The planning for sampling in PQ did not include such constraints, and more transects were used in PQ such that some transects could have been placed in non-coral areas. It probably helped explain why estimates by LIT were substantially higher than the later method.

There is actually a view that LIT considerably underestimates coral coverage compared to PQ. One study argued that apparent underestimates by LIT arose from using shorter transects (10 m) whereas overestimates by LIT or LPT reported in other studies had used longer transects (> 10 m), which can easily conform to the benthic substrate (Nakajima et al., 2010). When we consider that monitoring in Danajon Bank used longer transects than 10 m, PQ might not have been the one to underestimate, but LIT could have overestimated coral cover in this study.

Furthermore, quadrat-based methods can underestimate coral cover (Leujak and Ormand, 2007; Jokiel et al., 2015; Sant et al., 2017). PQ involves determination of substrate cover on screen images which often suffer from limited resolution and contrast (Leujak and Ormand 2007). This makes it difficult or impossible to identify substrate types, especially if they are in shade. Indeed, the problem of image resolution is pronounced underwater even if the digital camera allows high resolution images. This is mainly because it is difficult for divers to hold the camera still, sand and silt often disturb the visibility of water as they swim, and again shade is a common problem (Leujak and Ormand, 2007). Difficulties in identifying accurate substrates leads observers to categorize them underneath random points as "shadow" and as a result, miss points which could have been coral.

Although subsampling did not help make LIT estimates comparable with PQ estimates in temporal patterns of coral cover, this research adds to evidence that precision can fluctuate with sample sizes. One simulation study discussed that the precision of estimates by LIT and LPT sharply diminished as coral cover declined below 15 % when 10 transects (20 m) were used (Nadon and Stirling, 2006). This research suggests that at least 5 ~ 10 transects are required to maintain precision and obtain significant inference with sufficient power depending on the percent coral cover. A second study, using the same PQ dataset as this study from 2007 and 2008, investigated the minimum sampling effort powerful enough to detect relative changes in coral cover with a power of 80 % at $\alpha = 0.05$ (Molloy et al., 2013). This simulation study discussed that the reliability to detect relative changes depends on sample sizes. Based on the power analysis, it suggests that optimal sample sizes would be 10 transects with 23 quadrats within each transect, and influenced collection of data used in this study. These studies showed that the precision of estimates would support the ability to detect small changes between years when such differences are real rather than sampling errors. However, it can be influenced by the reef patchiness as well as coral cover (Brown et al., 2004; Nadon and Stirling, 2006) such that required sampling effort depends on characteristics of coral reefs at study sites.

The result of the temporal patterns emphasized the need for statistically powerful sample sizes. It leads to a suggestion that we better follow the recommendation of using all 230 quadrats to maintain the statistical power of PQ data. This is especially so given that using PQ 230 data did not exhibit apparent spatial autocorrelation, and that the imbalance of sample sizes between LIT and PQ did not violate the assumption of equal variance across years in models. However, we have to note that coral cover ranged from 1 ~ 60 % in our MPAs (Figs. I.1 ~ 9) and thus, the precision of estimates might vary across sites although the extent to which precision varied is unknown.

As mentioned above, subsampling LIT data did not make data from LIT and PQ comparable in estimated temporal patterns of coral cover. Possible reasons for this result are twofold. First, sample sizes of LIT point data (n = 40 per site) were still not large enough to estimate precise patterns of coral cover. It is possible that coral cover had not changed over years at all such that a result of no significant trend was actually true. However, given that PQ data with larger sample sizes (i.e. PQ 230 data: sample n = 230 per site) were likely to produce more accurate estimates, a difference in results between LIT point data and PQ data, specifically PQ 230 data, indicates that the former needed larger sample sizes. Therefore, adding deep transect data should be considered if there was no significant depth effect on coral between shallow (< 5 m) and deep (5 ~ 10 m) zones. Second, PQ 40 data detected significant trends while LIT point data did not in some MPAs despite the same sample sizes (sample n = 40 per site). This contradiction leads to the next possible reason; a difference in the coverage of study sites between LIT and PQ. The spatial extent of data sampling can influence results along with sample sizes (Dungan et al., 2002), and it seems to have been the case in the results here as well; LIT used only two 20 m transects while PQ had ten 23 m transects deployed at each study site. Even if the sample sizes were the same for LIT point data and PQ 40 data in the number of quadrats, the latter was sourced from PQ and hence covered wider ranges of MPAs. This fact might explain the result that subsampling LIT data did not make apparent changes in LIT point data.

Providing that estimated temporal patterns from the two methods were comparable, one question will arise; which data should we use for years when both the old and new methods were conducted? Three options are: (1) to use data from the old method only; (2) to use data from the new method only; and (3) to use all data points from both methods. The best option is likely to be the second one given that one method was conducted following the other in the same site. First, using data from both methods would result in overlapping transects. Since monitoring data rely upon each replicated sample not associated with the other replicates (Hill and Wilkinson, 2004), we should avoid using combined data with possible pseudoreplication. Second, estimates from larger sample sizes are likely to be more precise. When the purpose of changing sampling methods in MPA monitoring is to increase sample sizes, using data from the later method would be better to increase the quality of assessment. Thus, for the future assessment of MPAs in this region, using only PQ data is recommended.

The analysis of semivariance helped ease the concern of spatial autocorrelation in PQ data. If semivariograms exhibit elevations of semivariance, data are correlated (Palmer and McGlinn, 2017). In the semivariograms here, however, there was no clear elevation of semivariance in most sites. This result suggests that coral in the study sites was distributed relatively evenly, and that placing photoquadrats at 1 m intervals along connected transects did not create substantial autocorrelation. The next step would be to investigate whether the imbalanced sample sizes between LIT and PQ data did not violate the assumption of homogeneity (equal data variance) when assessing the effect of MPAs. If this condition was met, the whole dataset should be used rather than subsampling PQ data to keep the statistical power. This suggestion can be further supported by the analysis of the temporal patterns of coral cover which showed that PQ 40 data lost the statistical power to some extent.

I demonstrated that subsampling data to increase sample sizes has some potential to improve comparability of various datasets if we must, although good design *ab initio* is far better than post hoc statistical remedies (see 3.2.3). Increasing sample sizes has several merits including improved accuracy and precision of estimates. Furthermore, it enhances statistical power to detect small changes, lowers the variability of data, and increases chances to meet model assumptions such as normality and equal variance (Lumley et al., 2002; Hernandez et al., 2006; Button et al., 2013). This study also confirmed that synthesising data from different sampling techniques requires intensive field survey knowledge and analytical skills. However, the results shown here led us to encouraging observations that when discrepancies between different methods are identified and offset, coral cover measured by these methods can effectively and legitimately be combined. This detailed investigation into integrating data from different methods provides a foundation for a grounded analysis of changes in coral cover over time and assessment of the long-term protection effect on coral.

Table 2.1 Characteristics of eight MPAs (from west to east) in Danajon Bank, where coral cover data analyzed in this study were collected

Site	**Year established**	**Age (as of 2016)**	**Area (ha)**	**Compactness**	**Distance from the nearest village (km)**	**Ecological zone**	**Initial % coral cover**
Bilang Bilagan	1999	17	10.5	0.41	0.63	Reef	36.1
Batasan	1999	17	21.0	0.67	0.65	Reef	46.1
Pandanon	2002	14	20.0	0.08	1.17	Reef	40.4
Asinan	2000	16	50.0	0.44	2.09	Reef	76.6
Jandayan Norte	2002	14	24.9	0.53	0.46	Coastal	22.4
Handumon	1995	21	50.0	0.51	0.18	Coastal	27.6
Pinamgo	2000	16	37.8	0.28	0.93	Reef	17.5
Bantiguian	2004	12	10.6	0.69	1.27	Coastal	27.8

Table 2.2 Years the line intercept transect (LIT), photoquadat (PQ), and both LIT and PQ surveys were conducted at eight MPAs. Each MPA has different survey years, but no survey was conducted in Dry 2007, Wet 2011, Wet 2015, and Wet 2016 at all MPAs.

Site	Years LIT surveys conducted	Years PQ surveys conducted	Years both surveys conducted
Bilang Bilangan	Wet 1998 ~ Wet 2010	Wet 2007 ~ Dry 2016	Wet 2008, Dry 2010, Wet 2010
Batasan	Dry 1999 ~ Wet 2010	Wet 2007 ~ Dry 2016	Wet 2008, Dry 2010, Wet 2010
Pandanon	Wet 1998 ~ Wet 2010	Wet 2007 ~ Dry 2016	Wet 2008, Dry 2010, Wet 2010
Asinan	Wet 1998 ~ Wet 2010	Dry 2008 ~ Dry 2016	Dry 2010, Wet 2010
Jandayan Norte	Dry 2003 ~ Wet 2010	Wet 2007 ~ Dry 2016	Wet 2008, Dry 2010, Wet 2010
Handumon	Dry 1999 ~ Wet 2010	Wet 2007 ~ Dry 2016	Wet 2008, Dry 2010, Wet 2010
Pinamgo	Wet 2002 ~ Wet 2010	Dry 2008 ~ Dry 2016	Dry 2009, Dry 2010, Wet 2010
Bantiguian	Dry 2004 ~ Wet 2010	Dry 2008 ~ Dry 2016	Dry 2010, Wet 2010

Table 2.3 Major benthic habitat categories, sub-categories, and the code of each sub-category. One sub-category (non-aquatic object: NAO) was added to Abiotic, and one major category (TWS) with three sub-categories (Tape, Wand, and Shadow) was included for the PQ survey.

Major category	Sub-category	Benthic code
Coral	Acropora	ACT
	Coral branching	CB
	Coral encrusting	CE
	Coral foliose	CF
	Coral massive	CM
	Coral mushroom	CMR
	Coral sub-massive	CSM
Dead coral	Dead coral	DC
	Dead coral algae	DCA
	Rubble	R
Other coral	Fire coral	CME
Octocoral	Gorgonian	GORG
	Soft coral	SC
Invertebrate	Anemone	ANE
	Ascidian	ASC
	Bivalve mollusk	CLAM
	Hydroid	HYD
	Sponge	SP
	Tube worm	TUBE
	Other invertebrates	OT
	Zoanthid	ZO
Algae	Algal assemblage	AA
	Coral algae	CA
	Halimeda	HA
	Macro-algae	MA
	Sargassum	SAR
	Turf algae	TA
Seagrass	Seagrass	SG
Abiotic	Rock	RCK
	Sand	SA
	Silt	SI

Table 2.4 Subsampling treatments produced from subsampling line intercept transect (LIT) and photoquadrat (PQ) data. √ indicates in which analyses (the Wilcoxon signed-rank test, or linear and linear mixed effects models) each treatment was analyzed.

Subsampling treatment	**Descriptions**	**Treatment analyzed**	
		Wilcoxon	**Linear**
LIT line data	Line intercept transect data before subsampling (sample n = 2)	√	√
LIT point data	Line intercept transect data after subsampling (sample n = 40)	√	√
PQ 230 data	Photoquadrat data with all number of quadrats (sample n = 230)	√	√
PQ 80 data	Photoquadrat data with 80 randomly selected quadrats (sample n = 80)	√	
PQ 40 data	Photoquadrat data with 40 randomly selected quadrats (sample n = 40)	√	√
LIT line + PQ 230 data	LIT line and PQ 230 combined data		√
LIT point + PQ 40 data	LIT point and PQ 40 combined data		√

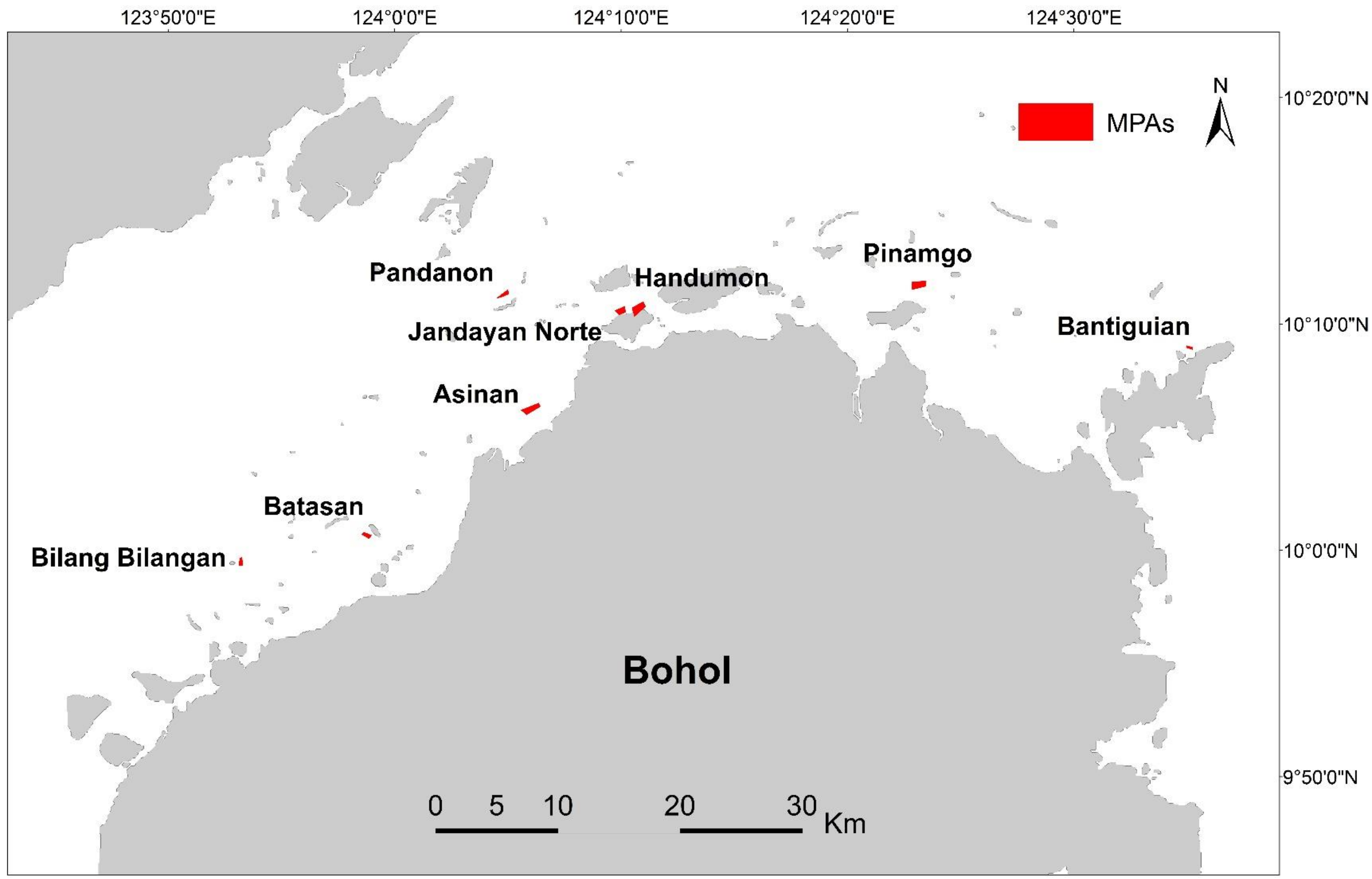

Fig. 2.1 Map of Bohol Island, the Philippines, indicating locations of eight MPAs on Danajon Bank

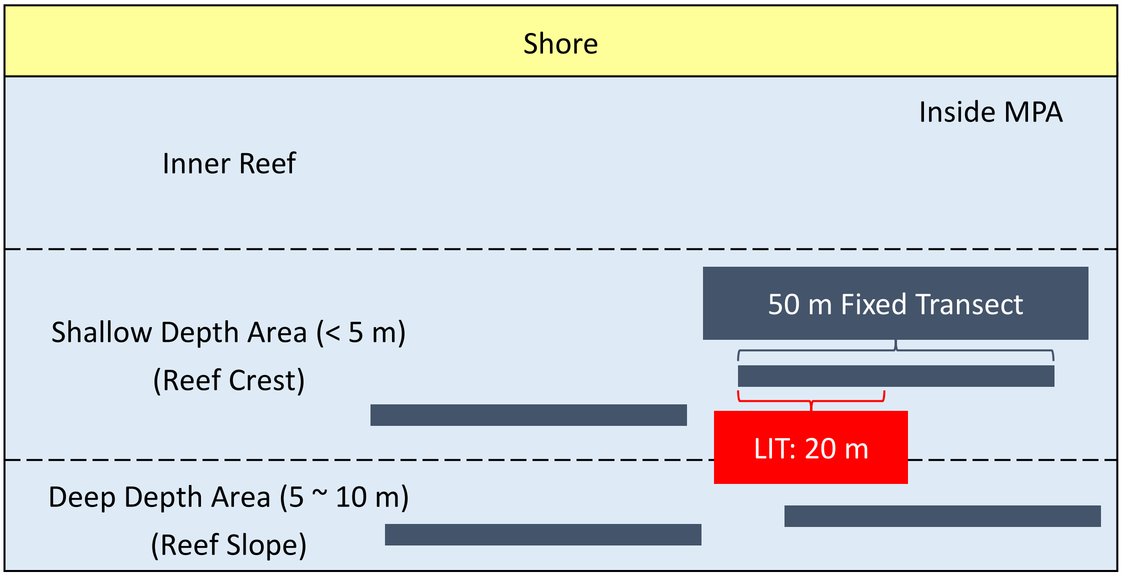

Fig. 2.2 Aerial view of the deployment of the fixed transects for the line intercept transect (LIT) survey inside the MPA. Other four transects were placed outside the MPA in the same way.

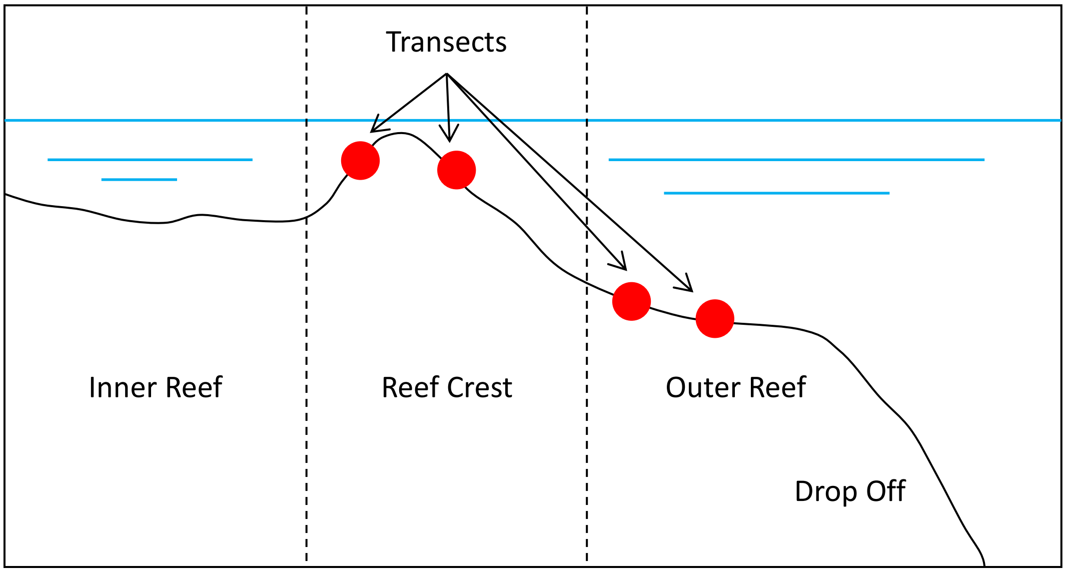

Fig. 2.3 Side view of the deployment of the fixed transects for the line intercept transect (LIT) survey.

Transects for inside and outside the MPA were placed in the same way, respectively.

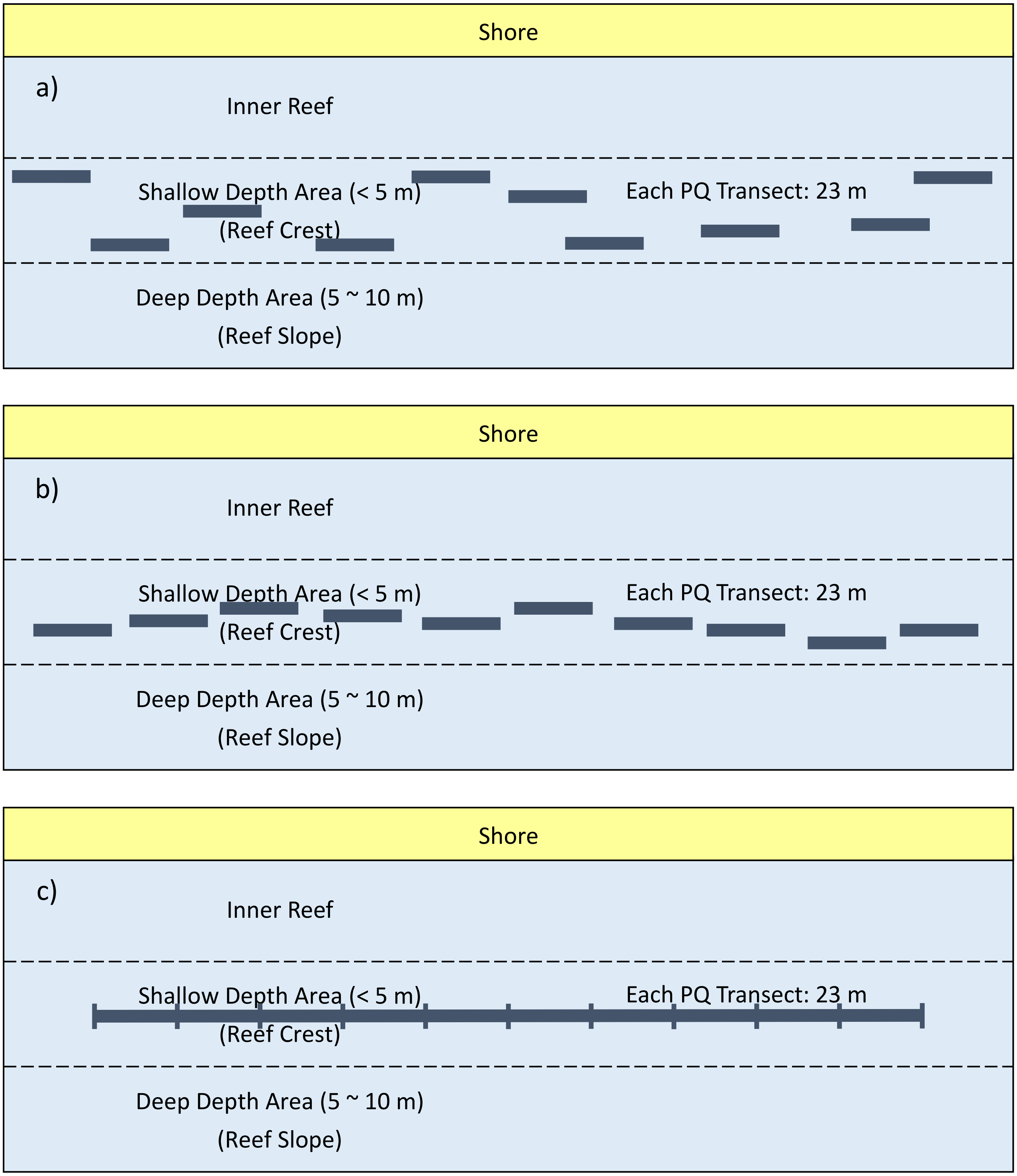

Fig. 2.4 Deployment of the PQ transects. (a) represents how PQ transect tapes were supposed to be placed (PQ transect tapes haphazardly placed). (b) represents how PQ transect tapes were placed from Wet 2007 onwards (PQ transect tapes placed along the same depth contour, but separately). (c) represents how PQ transect tapes were placed as of Dry 2016 (PQ transect tapes placed end to end, each of which was 25 ~ 26 m to make 2 ~ 3 m intervals between transects). When the tapes were first connected was not documented and thus, not identified.

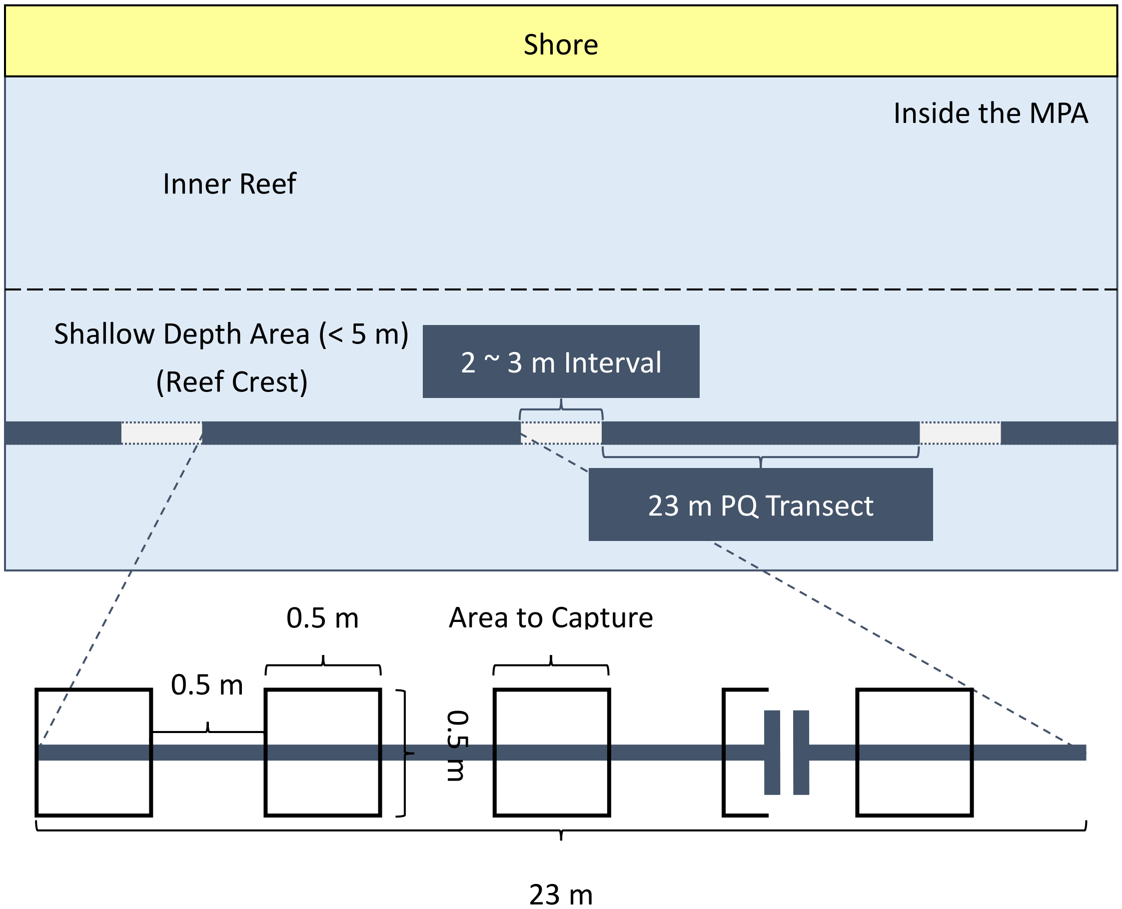

Fig. 2.5 Deployment of the transect tapes used for the photoquadrat (PQ) survey as of Dry 2016

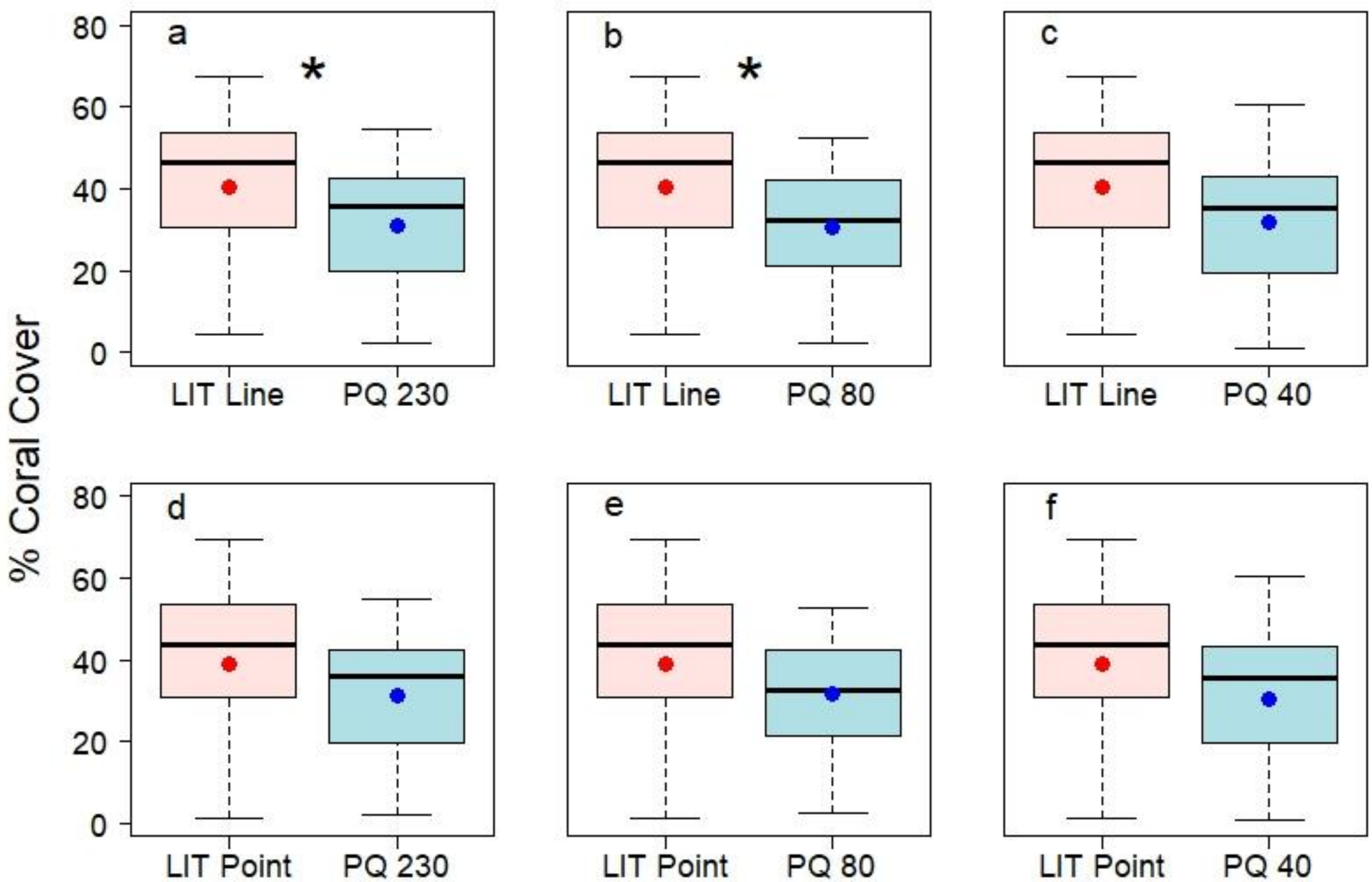
% Coral Cover
80
60
40
20
0
a
*
LIT Line
PQ 230
b
*
LIT Line
PQ 80
c
LIT Line
PQ 40
d
LIT Point
PQ 230
e
LIT Point
PQ 80
f
LIT Point
PQ 40

Fig. 2.6 Differences in mean percent coral cover between line intercept transect (LIT) and photoquadrat (PQ) data (Wet 2008 ~ Wet 2010). Six plots are comparisons between (a) LIT line and PQ 230 data, (b) LIT line and PQ 80 data, (c) LIT line and PQ 40 data, (d) LIT point and PQ 230 data, (e) LIT point and PQ 80 data, and (f) LIT point and PQ 40 data. LIT line and LIT point data indicate LIT data before and after subsampling, respectively. PQ 230, PQ 80, and PQ 40 data indicate PQ data with all quadrats (sample n = 230), 80 randomly selected quadrats, and 40 randomly selected quadrats, respectively. Dots in the boxes indicate mean values, and bold horizontal lines indicate median values. Significant differences between the two methods are indicated by asterisks (using a Bonferroni corrected p-value = 0.0083). Each subsampling treatment consists of 22 values, each of which is a mean of data in one sampling survey at one of eight MPAs. However, each mean was calculated based on a different number of observations, depending on which treatment was used: (1) LIT line data: sample n = 2; (2) LIT point data: sample n = 40; (3) PQ 230 data: sample n = 230; (4) PQ 80 data: sample n = 80; and PQ 40 data: sample n = 40.

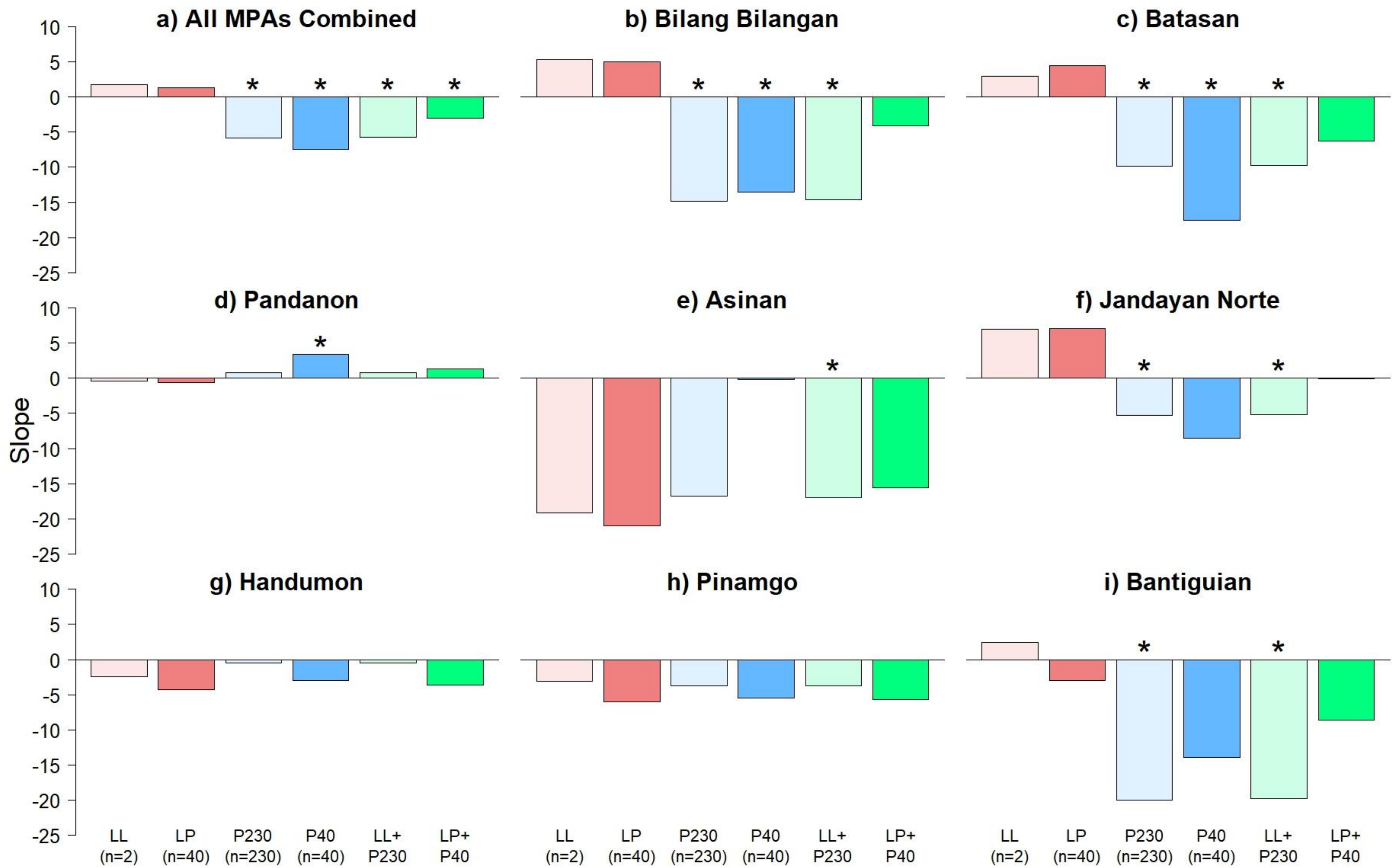
a) All MPAs Combined
b) Bilang Bilangan
c) Batasan
d) Pandanon
e) Asinan
f) Jandayan Norte
g) Handumon
h) Pinamgo
i) Bantiguian
Slope
10
5
0
-5
-10
-15
-20
-25
LL (n=2)
LP (n=40)
P230 (n=230)
P40 (n=40)
LL+ P230
LP+ P40
*

Fig. 2.7 Slopes estimated by the linear mixed effects models (a: all MPAs combined) and the linear models (b - i: each individual MPA laid from west to east) indicating temporal patterns of percent coral cover (Wet 2008 ~ Wet 2010) using the following sampling treatments: (1) LL – LIT line data; (2) LP – LIT point data; (3) P230 – PQ 230 data; (4) P40 – PQ 40 data; (5) LL+P230 – LIT line + PQ 230 data; and (6) LP+P40 – LIT point + PQ 40 data. Positive slopes indicate increases in percent coral cover and negative slopes decreases in percent coral cover. Significant temporal patterns (significant slopes) are shown with asterisks (using a Bonferroni corrected p-value = 0.0083). Treatments used in the linear mixed effects models and linear models differed in the number of observations, and models were based on all observations rather than mean values. Parentheses under the labels of subsampling treatments show the number of observations (sample n).

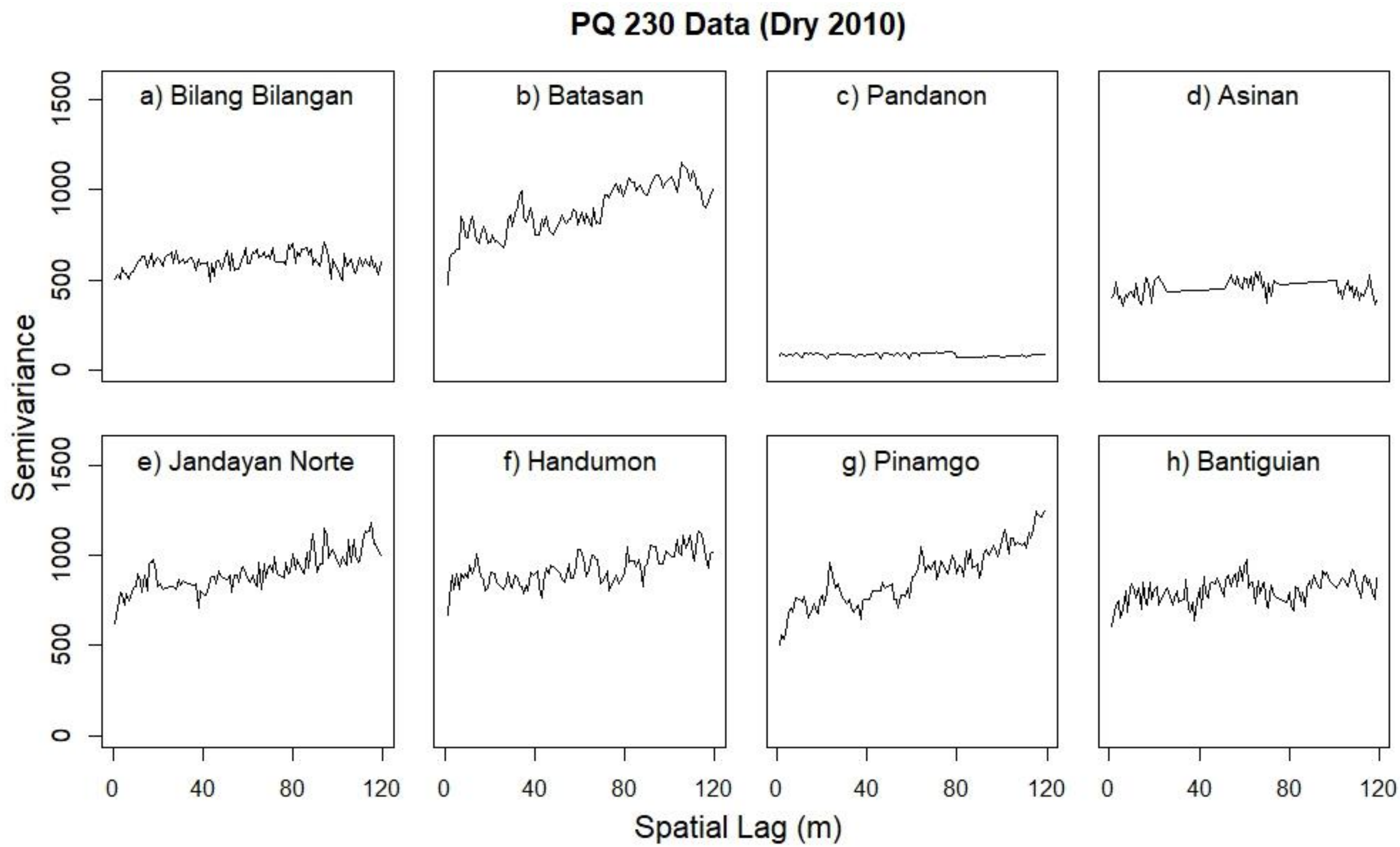

Fig. 2.8 Semivariograms to visually inspect spatial autocorrelation of PQ 230 data (photoquadrat data with all quadrats: sample n = 230) collected in the dry season of the year 2010. Eight MPAs where data were collected are laid from west to east.

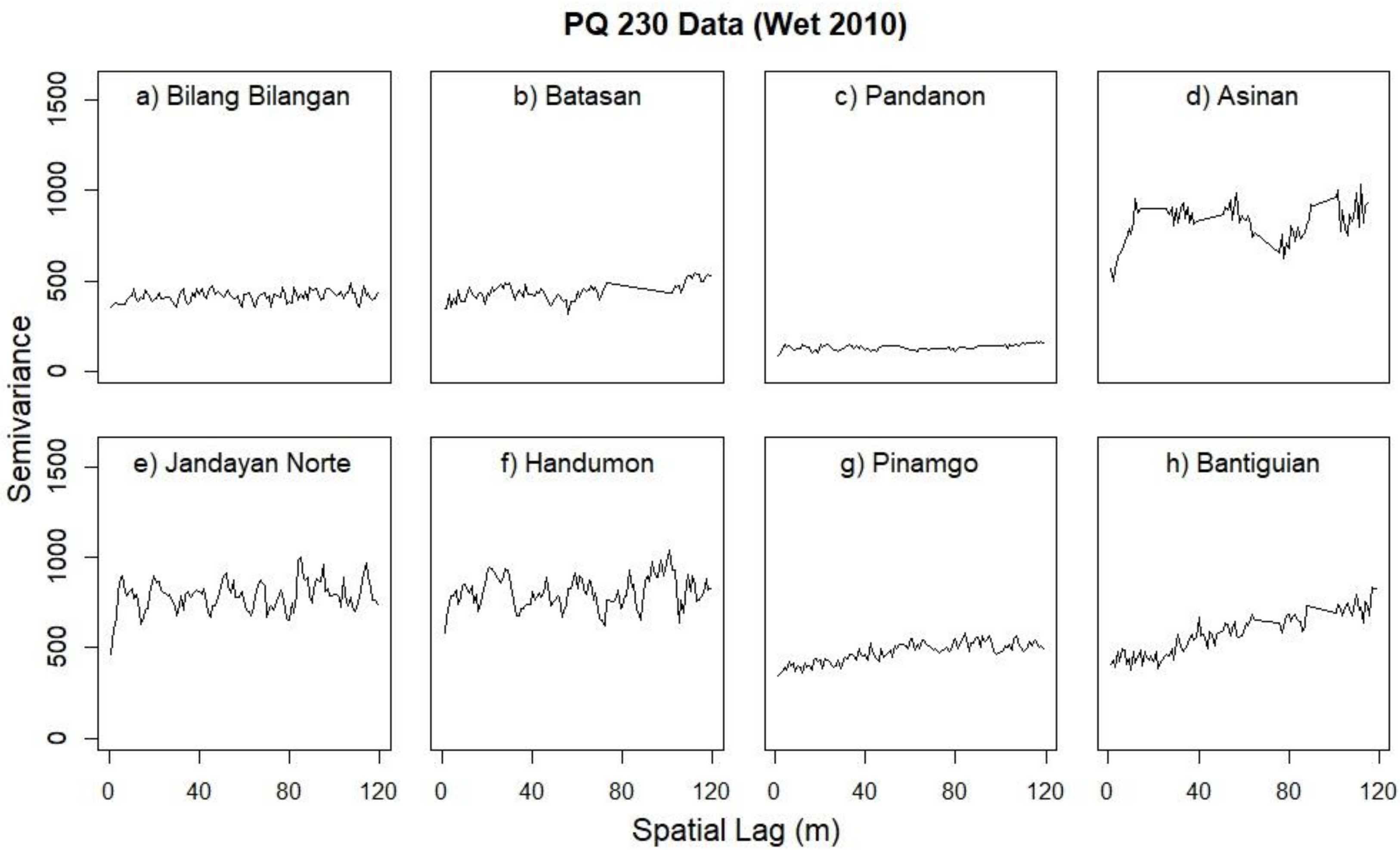

Fig. 2.9 Semivariograms to visually inspect spatial autocorrelation of PQ 230 data (photoquadrat data with all quadrats: sample n = 230) collected in the wet season of the year 2010. Eight MPAs where data were collected are laid from west to east.

Chapter 3: Conclusion

3.1 Overall conclusion

This study emphasizes that evaluating the long-term effect of habitat protection on coral is very complicated, especially when monitoring involves a sequential use of multiple methods. When data from different methods, such as LIT and PQ used in this study, are combined and analyzed, discrepancies between the two methods can be influenced by errors resulting from differences in accuracy and precision. In the context of benthic cover assessment, accuracy is defined as the ability to estimate true values and precision as the ability to detect small changes (Leujack and Ormand, 2007). Both are important for evaluation of change, whether from threats or from protection. In situations where data originate from multiple sources, comparisons of estimates are critical to confirm whether the methods are compatible enough to integrate into a single analysis of habitat change.

My research demonstrated that subsampling the LIT dataset provided a practical and effective way to combine datasets from the two sampling methods. In the comparative study of LIT and PQ, I found that the former tended to estimate higher mean percent coral cover than the latter, and that subsampling LIT data made it comparable with PQ data in estimated values. This finding indicates that large differences in sample sizes between the methods can prevent accurate data comparison.

I also found that LIT estimated no temporal patterns of percent coral cover whereas PQ detected significant decreases, and that subsampling LIT data actually did not help make temporal patterns estimated by the two methods comparable. In keeping with common statistical results, my finding confirmed that the power to detect statistical significance depends on sample sizes and the spatial extent of sampling (Dungan et al., 2002; Hernandez et al., 2006; Button et al., 2013). It emphasizes the importance of large sample sizes covering wide areas of the study sites for precise estimates.

3.2 MPA long-term monitoring: learning from the past

3.2.1 Benefits and challenges of changing sampling methods

Knowing the true nature and extent of coral cover is usually challenging (Nadon and Stirling, 2006), making it difficult to conclude which method is the most accurate. However, changing the sampling method can provide several benefits such as increases in sample sizes, which can contribute to the accuracy of data. Since the PQ method records data using digital recording devices, it enables efficient sampling and hence allows us to collect more information within a given time frame. Thus, PQ is likely to represent more accurate values than the other methods in which benthic coverage are measured *in situ* (Leujak and Ormand, 2007; Van Rein et al., 2011; Trygonis and Sini, 2012). Findings in this study, combined with results from other research, suggest that PQ is more appropriate for estimating coral cover and for informing local communities of the status of coral reefs.

Since the PQ method employs photography, it provides three additional benefits. First, the digital images collected using this method provides a semi-permanent record, allowing scientists to ask additional questions in future studies (Preskitt et al., 2004). Second, PQ can improve the accuracy of data by allowing analysts to confirm the benthic coverage and identification multiple times (Mantelatto et al., 2013). Third, PQ data are not affected by observer's bias unlike LIT where data are quantified *in situ* (Šaškov et al., 2015). Thus, the PQ method allows any certified diver to participate in monitoring once they have been sufficiently trained to conduct underwater photography.

Although efficient in the field, the PQ method can bring other costs. Processing PQ images involves computer time, adding lab-based labour costs (Molloy et al., 2013). Depending on the scale of surveys, PQ photo analyses can be labour-intensive. In this case, image analyses might necessitate extra expenses to employ more personnel to process data. It is thus critical to identify an efficient sampling design which balances the sampling costs and processing time (Molloy et al., 2013).

The most challenging part of changing sampling methods arises when data from multiple methods are combined into one time-series dataset. First, the new method has to be designed so that it becomes compatible with the former method (e.g. sample units such as quadrats or transects, spatial extent, etc.). For instance, the LIT survey in this study was conducted using shallow and deep transects whereas the PQ survey was conducted only at the shallow depth. However, coral has a depth-dependent distribution (Bridge et al., 2013; Guest et al., 2016). Therefore, sampling at different depths may influence the results. Second, the comparative assessment of the old and new methods should ideally be conducted during the method transition years. Such timing would allow for a rapid method adjustment based on data variability between the two methods and would improve the capacity of monitoring to detect long-term trends. Third, another difficulty is the lack of a clear approach to determine the length of time for method comparisons. In the case of monitoring at Danajon Bank, the short method transition period when both methods were used (Wet 2008 ~ Wet 2010) could have been insufficient for comparing temporal patterns. In addition, the study sites were not surveyed for every sampling survey during the transition years, probably making comparisons less robust (Figs. I.1 ~ 9). Given the possible variability of data or sampling errors, a longer time period might have been necessary to assess the estimate of temporal patterns.

3.2.2 Improving MPA long-term monitoring: future challenges

A major challenge of monitoring is the need to obtain precise monitoring data which allows for detection of significant changes across space and time (Schmeller et al., 2009). The quality of data should be carefully maintained, especially when different methods were used in monitoring, but this need can often conflict with logistical constraints, including finite financial and human resources (Schmeller et al., 2009). Long-term monitoring in Danajon Bank faced three major difficulties.

First, inadequate exchange between the project scientists and field biologists led to poor adherence to procedures that were designed to provide statistical power. The field biologists adapted sampling methods to optimise logistical efficiency rather than strictly following survey procedures. For example, it seems that the field biologists began to connect transect tapes during the PQ survey into one long line to enable quick sampling (Fig. 2.4). Placing samples closer together, indeed sequentially linking them, increased the risk of spatial autocorrelation and might have reduced the representativeness of samples (Underwood, 1997). In this case, placing connected transects in a single line along the reef crest might also have limited the capacity of sampling to document trends in other reef zones. This sort of challenge indicates the need for the project scientists to emphasize adherence to protocols, visit the field sites frequently, and query procedures insistently. The field biologists also have obligations to follow protocols or query them openly, and required to discuss with the scientists when sampling protocols have to be changed to ensure proper alterations.

The second difficulty with monitoring in Danajon Bank arose around the accuracy of data encoding. Data related-problems largely derived from three sources: misspelling or mislabeling during data entry, missing information, and data arrangement in spreadsheets that was incompatible with analyses (Appendix B). In addition, inconsistencies in benthic image labeling affected PQ data. These inconsistencies were possibly due to inconsistent nomenclature standards used by encoders. Such errors were multiplied by a large volume of PQ data. I addressed data errors in several steps (see Appendices B and D for more details). To clean data, I first obtained original data files (LIT: spreadsheets, PQ: digital images, CPCe files, and spreadsheets), all of which were stored in the database of Project Seahorse at the University of British Columbia. Second, I identified missing data by sorting out files by site, year, season, and transect, and stored sorted files in a newly set database. Third, reasons for missing data were confirmed with the biologists at Zoological Society of London

- Philippines (ZSL - Philippines). If data were found at ZSL - Philippines, they were added to the database for the sorted files; if surveys were not conducted, data were treated as NAs. Fourth, I made new data spreadsheets in a csv format based on original data. This process was needed to import data files into R statistical software for analyses because original data were not in an appropriate format. In the format translation, I arranged information within each file chronologically and numerically, and corrected misspelling or mislabeling of data (e.g. incorrect benthic codes, mislabeled photoquadrat information). Error correction also required confirming whether each error was corrected accurately. Database accuracy was a major challenge in this study, and data cleaning comprised the core part of pre-analysis time. In circumstances where data encoding is complicated, clear guidelines should be provided by the filed biologists to standardize data entry systems, specifically when different volunteers are employed in every monitoring survey.

Third, one consequence of monitoring challenges – both in managing and analyzing data – has been limited communication of trends in MPAs with local communities, who primarily mange MPAs. The obvious solution to this problem would be to assign someone to analyze data, advise the field biologists, and provide regular feedbacks to local communities. One possible way would be to regularly have a graduate student or postdoc in charge of these tasks as part of their research. However, as I have indicated, analyzing these data is hugely challenging. Limited funding also makes this option difficult. A second solution would be to increase the involvement of local villagers or fishers in monitoring. In theory, having primary stakeholders (i.e. local communities) participate in monitoring can help identify or confirm site-specific needs for management actions and prompt management responses (Lundquist and Granek, 2005; Ferse et al., 2010). However, participatory monitoring has often encountered genuine limitations, such as failure to follow protocols and poor documentation, which can be affected by motivation of local communities (Heck et al., 2011).

Lastly, to further improve the long-term monitoring, revising the current sampling scheme would be worth considering. In a case of the PQ survey at Danajon Bank, systematic sampling would be more appropriate for the future monitoring given the past sampling method. More specifically, one 230 m transect should be placed randomly for each site rather than pretending to place ten 23 m transects with 2 ~ 3 m intervals between. Benthic images should still be taken systematically at 1 m intervals as was done previously, and quadrats should be retained as the sample unit. In this way, the problem of pseudoreplication can be avoided while the sampling effort suggested by Molloy et al. (2013) can be maintained. In addition, it allows time-efficient sampling. Furthermore, placing another transect in the deep area (5 ~ 10 m) would be helpful even for the PQ survey to see how different depths affect the estimate by PQ data. For data management, the image labeling system has to be established using the coherent standardized site, location, transect, and quadrat codes (Table E.5). Most importantly, the quadrat code has to begin with 0 such that each benthic image is labeled as 01, 02, 03, 04, …, 23. The label with leading zeros allows Excel to arrange quadrat information numerically whereas quadrats labeled without 0 will be ordered as 1, 10, 11, 12, …, 2, 21, 22, 23. It will make data management significantly easier and hence minimize data encoding errors.

3.2.3 For the future MPA long-term monitoring programs

This study highlighted that subsampling can be a useful statistical approach to help make datasets from different sampling methods comparable. However, we have to note that it is merely a remedy for previously collected data, and is not a panacea to solve egregious problems with data collection. Our LIT data were sourced from unreliably fixed transects to an extent that we have here treated the transects as haphazard rather than fixed. Resulting low sample sizes were improved by subsampling LIT data, yet such subsampling carried its own risks given that transects may have overlapped enough to reduce independence.

A series of challenges we faced in this research, including subsampling, provide us with four recommendations for future MPA long-term monitoring. First, surveyors should randomly sample data rather than using fixed transects or quadrats. The difficulties we faced in trying to maintain the fixed transects are likely to occur in other coral reef sites. Second, a preliminary survey should be conducted to ensure adequate sample sizes as was done by Molloy et al. (2013). Since the power to detect change is influenced by coral cover and reef patchiness (Brown et al., 2004; Nadon and Stirling, 2006), the appropriate sample sizes should be estimated in advance for each study site and monitoring. Third, the survey should cover as wide areas of study sites as possible within a given time restricted by the SCUBA tank air. There are reefs in Danajon Bank, where distribution of coral is uneven across sites (author's observation); thus, data from the portion of the MPA might not represent the coral cover of the whole area. Fourth, a stratified sampling design should be implemented to proportionally allocate sampling effort within each study site (Krebs, 1999; Nadon and Stirling, 2006). MPAs on Danajon Bank vary in area and coral cover. Therefore, deciding sample sizes according to MPA area and coral cover could help study sites receive proper representation. These four recommendations can also be applied to the introduction of new sampling methods in ongoing MPA monitoring.

3.3 Moving forward to better manage MPAs

MPAs play an important role in marine conservation, and the effectiveness of their protection can be enhanced with the information garnered through accurate long-term monitoring. Long-term evaluations can confirm whether MPAs meet their stated conservation objectives, identify issues confronting MPAs, and support solutions to emergent issues based on scientific data. Understanding how sampling techniques influence data generation is essential to correctly interpret the habitat protection effect of MPAs. It is especially important in circumstances where the method was changed in the middle of monitoring (often with an intention of remedying past deficiencies in sampling). By identifying an approach for integrating the two sampling methods, my research provides guidance for accurately monitoring habitat change over time. Assessing the influence of methods on detecting change is vital for the successful monitoring survey and can help enhance marine conservation.

References

Abesamis, R. A., & Russ, G. R. (2005). Density‐dependent spillover from a marine reserve: Long‐term evidence. Ecological applications, 15 (5), 1798-1812.

Aburto-Oropeza, O., Erisman, B., Galland, G. R., Mascareñas-Osorio, I., Sala, E., & Ezcurra, E. (2011). Large recovery of fish biomass in a no-take marine reserve. PLoS One, 6 (8), e23601.

Addison, P. (2011). A global review of long-term Marine Protected Area monitoring programmes: the application of a good framework to marine biological monitoring.

Alcala, A. C., Russ, G. R., Maypa, A. P., & Calumpong, H. P. (2005). A long-term, spatially replicated experimental test of the effect of marine reserves on local fish yields. Canadian Journal of Fisheries and Aquatic Sciences, 62 (1), 98-108.

Allison, G. W., Lubchenco, J., & Carr, M. H. (1998). Marine reserves are necessary but not sufficient for marine conservation. Ecological applications, 8 (sp1).

Andrew, N. L., & Mapstone, B. D. (1987). Sampling and the description of spatial pattern in marine ecology. Oceanography and Marine Biology, 25, 39-90.

Armstrong, R. A. (2014). When to use the Bonferroni correction. Ophthalmic and Physiological Optics, 34 (5), 502-508.

Ateweberhan, M., McClanahan, T. R., Graham, N. A. J., & Sheppard, C. R. C. (2011). Episodic heterogeneous decline and recovery of coral cover in the Indian Ocean. Coral Reefs, 30 (3), 739.

Baldwin, K, Punnett, S, & Smith, C. (2005). Involving communities in the implementation of Reef Check: Strategies for co-management of marine resources in St. Vincent and the Grenadines.

Barrett, N. S., Edgar, G. J., Buxton, C. D., & Haddon, M. (2007). Changes in fish assemblages following 10 years of protection in Tasmanian marine protected areas. Journal of Experimental

Marine Biology and Ecology, 345 (2), 141-157.

Beard, G. R., Scott, W. A., & Adamson, J. K. (1999). The value of consistent methodology in long-term environmental monitoring. Environmental Monitoring and Assessment, 54 (3), 239-258.

Bellwood, D. R., Hoey, A. S., & Hughes, T. P. (2011). Human activity selectively impacts the ecosystem roles of parrotfishes on coral reefs. Proceedings of the Royal Society of London B: Biological Sciences, rspb20111906.

Béné, C., Barange, M., Subasinghe, R., Pinstrup-Andersen, P., Merino, G., Hemre, G. I., & Williams, M. (2015). Feeding 9 billion by 2050–Putting fish back on the menu. Food Security, 7 (2), 261-274.

Benjamini, Y., & Yekutieli, D. (2001). The control of the false discovery rate in multiple testing under dependency. Annals of statistics, 1165-1188.

Biggerstaff, A., Jompa, J., & Bell, J. J. (2017). Increasing benthic dominance of the phototrophic sponge Lamellodysidea herbacea on a sedimented reef within the Coral Triangle. Marine Biology, 164 (12), 220.

Bowker, M. A., Muñoz, A., Martinez, T., & Lau, M. K. (2012). Rare drought-induced mortality of juniper is enhanced by edaphic stressors and influenced by stand density. Journal of Arid Environments, 76, 9-16.

Bremer, L. L., & Farley, K. A. (2010). Does plantation forestry restore biodiversity or create green deserts? A synthesis of the effects of land-use transitions on plant species richness. Biodiversity and Conservation, 19 (14), 3893-3915.

Bridge, T. C., Hughes, T. P., Guinotte, J. M., & Bongaerts, P. (2013). Call to protect all coral reefs. Nature Climate Change, 3 (6), 528.

Brown, E. K., Cox, E., Jokiel, P. L., Rodgers, S. K. U., Smith, W. R., Tissot, B. N., ... & Hultquist, J.

(2004). Development of benthic sampling methods for the Coral Reef Assessment and Monitoring Program (CRAMP) in Hawai'i. Pacific Science, 58 (2), 145-158.

Button, K. S., Ioannidis, J. P., Mokrysz, C., Nosek, B. A., Flint, J., Robinson, E. S., & Munafò, M. R. (2013). Power failure: why small sample size undermines the reliability of neuroscience. Nature Reviews Neuroscience, 14 (5), 365-376.

Cabral, R. B., Aliño, P. M., Balingit, A. C. M., Alis, C. M., Arceo, H. O., Nañola Jr, C. L., ... & Partners, M. S. N. (2014). The Philippine marine protected area (MPA) database. Philipp Sci Lett, 7 (2), 300-308.

Chirico, A. A., McClanahan, T. R., & Eklöf, J. S. (2017). Community-and government-managed marine protected areas increase fish size, biomass and potential value. PloS one, 12 (8), e0182342.

Christie, P., Armada, N. B., White, A. T., Gulayan, A. M., & de Dios, H. H. (2006). Coastal environmental and fisheries profile of Danajon Bank, Bohol, Philippines.

Cinner, J., McClanahan, T. R, Wamukota, A., Darling, E., Humphries, A., Hicks, C., ... & Bodin, Ö. (2013). Social-ecological vulnerability of coral reef fisheries to climatic shocks. FAO Fisheries and Aquaculture Circular (FAO) eng no. 1082.

Coglianese, C. (2012). Measuring Regulatory Performance evaluating the impact of regulation and regulatory policy.

Collard, S., Le Brocque, A., & Zammit, C. (2009). Bird assemblages in fragmented agricultural landscapes: the role of small brigalow remnants and adjoining land uses. Biodiversity and Conservation, 18 (6), 1649-1670.

Conway, C. J., & Gibbs, J. P. (2001). Factors influencing detection probability and the benefits of call broadcast surveys for monitoring marsh birds. Final Report, USGS Patuxent Wildlife Research

Center, Laurel, MD.

Copper, P. (1994). Ancient reef ecosystem expansion and collapse. Coral reefs, 13 (1), 3-11.

Côté, I. M., & Darling, E. S. (2010). Rethinking ecosystem resilience in the face of climate change. PLoS biology, 8 (7), e1000438.

Cox, C., Valdivia, A., McField, M., Castillo, K., & Bruno, J. F. (2017). Establishment of marine protected areas alone does not restore coral reef communities in Belize. Marine Ecology Progress Series, 563, 65-79.

De Santo, E. M. (2013). Missing marine protected area (MPA) targets: how the push for quantity over quality undermines sustainability and social justice. Journal of environmental management, 124, 137-146.

Diaz-Pulido, G., McCook, L. J., Dove, S., Berkelmans, R., Roff, G., Kline, D. I., ... & Hoegh-Guldberg, O. (2009). Doom and boom on a resilient reef: climate change, algal overgrowth and coral recovery. PLoS One, 4 (4), e5239.

Done, T. J., Ogden, J. C., Wiebe, W. J., & Rosen, B. R. (1996). Biodiversity and ecosystem function of coral reefs. In: Mooney, H. A., Cushman, J. H., Medina, E., Sala, O. E., and Schulze, E. D. (Eds.), Functional Roles of Biodiversity: A Global Perspective. John Wiley and Sons.

Donham, E., Foster, M. S., Rice, M. R., Cailliet, G. M., Yoklavich, M. M., & Hamilton, S. L. (2017). Natural History Observations of Hawaiian Garden Eels, Gorgasia hawaiiensis (Congridae: Heterocongrinae), from the Island of Hawai ‘i. Pacific Science, 71 (2), 135-147.

Dungan, J. L., Perry, J. N., Dale, M. R. T., Legendre, P., Citron ‐ Pousty, S., Fortin, M. J., ... & Rosenberg, M. (2002). A balanced view of scale in spatial statistical analysis. Ecography, 25 (5), 626-640.

English, S. S., Wilkinson, C. C., & Baker, V. V. (1997). Survey manual for tropical marine resources. Australian Institute of Marine Science.

Fabricius, K. E. (2005). Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. Marine pollution bulletin, 50 (2), 125-146.

Facon, M., Pinault, M., Obura, D., Pioch, S., Pothin, K., Bigot, L., ... & Quod, J. P. (2016). A comparative study of the accuracy and effectiveness of line and point intercept transect methods for coral reef monitoring in the southwestern Indian Ocean islands. Ecological indicators, 60, 1045-1055.

Ferse, S. C., Costa, M. M., Manez, K. S., Adhuri, D. S., & Glaser, M. (2010). Allies, not aliens: increasing the role of local communities in marine protected area implementation. Environmental Conservation, 37 (1), 23-34.

Fidler, R. Y., Turingan, R. G., White, A. T., & Alava, M. N. (2017). Reef-wide beneficial shifts in fish population structure following establishment of marine protected areas in Philippine coral reefs. Marine Ecology Progress Series, 570, 187-202.

Gałecki, A., & Burzykowski, T. (2013). Fitting Linear Models with Homogeneous Variance: The lm () and gls () Functions. In Linear Mixed-Effects Models Using R (pp. 89-111). Springer New York.

Gratwicke, B., & Speight, M. R. (2005). The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. Journal of fish biology, 66 (3), 650-667.

Green, R. H., & Smith, S. R. (1997). Sample program design and environmental impact assessment on coral reefs. In Proc. 8th Int. Coral Reef Symp (Vol. 2, pp. 1459-1464).

Green, S. J., Alexander, R. D., Gulayan, A. M., Migrinio III, C. C., Jarantilla-Paler, J., & Courtney,

C. A. (2002). Bohol island: its coastal environment profile. Cebu City, Philippines: Bohol Environment Management Office, Bohol and Coastal Resource Management Project.

Guest, J. R., Tun, K., Low, J., Vergés, A., Marzinelli, E. M., Campbell, A. H., ... & Steinberg, P. D. (2016). 27 years of benthic and coral community dynamics on turbid, highly urbanised reefs off Singapore. Scientific reports, 6.

Hackerott, S., Valdivia, A., Green, S. J., Côté, I. M., Cox, C. E., Akins, L., ... & Bruno, J. F. (2013). Native predators do not influence invasion success of Pacific lionfish on Caribbean reefs. PLoS one, 8 (7), e68259.

Halpern, B. S. (2003). The impact of marine reserves: do reserves work and does reserve size matter? Ecological applications, S117-S137.

Hansen, G. J., Ban, N. C., Jones, M. L., Kaufman, L., Panes, H. M., Yasué, M., & Vincent, A. C. (2011). Hindsight in marine protected area selection: A comparison of ecological representation arising from opportunistic and systematic approaches. Biological Conservation, 144 (6), 1866-1875.

Heck, N., Dearden, P., McDonald, A., & Carver, S. (2011). Stakeholder opinions on the assessment of MPA effectiveness and their interests to participate at Pacific Rim National Park Reserve, Canada. Environmental management, 47 (4), 603-616.

Hernandez, P. A., Graham, C. H., Master, L. L., & Albert, D. L. (2006). The effect of sample size and species characteristics on performance of different species distribution modeling methods. Ecography, 29 (5), 773-785.

Hill, J., & Wilkinson, C. L. I. V. E. (2004). Methods for ecological monitoring of coral reefs. Australian Institute of Marine Science, Townsville, 117.

Hughes, T. P., Huang, H. U. I., & Young, M. A. (2013). The wicked problem of China's disappearing coral reefs. Conservation Biology, 27 (2), 261-269.

Hughes, T. P., Rodrigues, M. J., Bellwood, D. R., Ceccarelli, D., Hoegh-Guldberg, O., McCook, L., ... & Willis, B. (2007). Phase shifts, herbivory, and the resilience of coral reefs to climate change. Current Biology, 17 (4), 360-365.

Jackson, J. B., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., ... & Hughes, T. P. (2001). Historical overfishing and the recent collapse of coastal ecosystems. science, 293 (5530), 629-637.

Jokiel, P. L., Rodgers, K. S., Brown, E. K., Kenyon, J. C., Aeby, G., Smith, W. R., & Farrell, F. (2015). Comparison of methods used to estimate coral cover in the Hawaiian Islands. PeerJ, 3, e954.

Jone, P. J. S., Qiu, W., & De Santo, E. M. (2011). Governing marine protected areas: getting the balance right (p. 126). United Nations Environment Programme (UNEP).

Jones, G. P., McCormick, M. I., Srinivasan, M., & Eagle, J. V. (2004). Coral decline threatens fish biodiversity in marine reserves. Proceedings of the National Academy of Sciences of the United States of America, 101 (21), 8251-8253.

Juffe-Bignoli, D., Burgess, N. D., Bingham, H., Belle, E. M. S., De Lima, M. G., Deguignet, M., ... & Eassom, A. (2014). Protected planet report 2014. Cambridge, UK: UNEP-WCMC.

Kaiser, M. J., Collie, J. S., Hall, S. J., Jennings, S., & Poiner, I. R. (2002). Modification of marine habitats by trawling activities: prognosis and solutions. Fish and Fisheries, 3 (2), 114-136.

Knowlton, N., Brainard, R. E., Fisher R., Moews M., Plaisance L., & M. Julian Caley, M. J. (2010). Coral reef biodiversity. In: Mclntyre A. D. (Ed.), Life in the World's Oceans: Diversity, Distribution, and Abundance (pp. 65-77). UK: Blackwell Publishing Ltd.

Koenig, C. C., Coleman, F. C., Grimes, C. B., Fitzhugh, G. R., Scanlon, K. M., Gledhill, C. T., & Grace, M. (2000). Protection of fish spawning habitat for the conservation of warm-temperate reef-fish fisheries of shelf-edge reefs of Florida. Bulletin of Marine Science, 66 (3), 593-616.

Kohler, K. E., & Gill, S. M. (2006). Coral Point Count with Excel extensions (CPCe): A Visual Basic program for the determination of coral and substrate coverage using random point count methodology. Computers & Geosciences, 32 (9), 1259-1269.

Krebs, C. J. (1999). Ecological Methodology., 2nd edn. Addison-Wesley Longman: Menlo Park, CA.

Kroon, F. J., Schaffelke, B., & Bartley, R. (2014). Informing policy to protect coastal coral reefs: Insight from a global review of reducing agricultural pollution to coastal ecosystems. Marine pollution bulletin, 85 (1), 33-41.

Kuhlmann, D. H. H. (1988). The sensitivity of coral reefs to environmental pollution. Ambio 17 (1), 13-21.

Lam, K., Shin, P. K., Bradbeer, R., Randall, D., Ku, K. K., Hodgson, P., & Cheung, S. G. (2006). A comparison of video and point intercept transect methods for monitoring subtropical coral communities. Journal of Experimental Marine Biology and Ecology, 333 (1), 115-128.

Leonard, G. H., & Clark, R. P. (1993). Point quadrat versus video transect estimates of the cover of benthic red algae. Marine Ecology Progress Series, 203-208.

Leujak, W., & Ormond, R. F. G. (2007). Comparative accuracy and efficiency of six coral community survey methods. Journal of Experimental Marine Biology and Ecology, 351 (1), 168-187.

Li, W., Goodchild, M. F., & Church, R. (2013). An efficient measure of compactness for two-dimensional shapes and its application in regionalization problems. International Journal of Geographical Information Science, 27 (6), 1227-1250.

Loya, Y. (1978). Plotless and transect methods. Coral reefs: research methods. UNESCO, Paris, 197-217.

Lubchenco, J., Palumbi, S. R., Gaines, S. D., & Andelman, S. (2003). Plugging a hole in the ocean:

the emerging science of marine reserves. Ecological applications, 13 (1), S3-S7.

Lumley, T., Diehr, P., Emerson, S., & Chen, L. (2002). The importance of the normality assumption in large public health data sets. Annual review of public health, 23 (1), 151-169.

Lundquist, C. J., & Granek, E. F. (2005). Strategies for successful marine conservation: integrating socioeconomic, political, and scientific factors. Conservation Biology, 19 (6), 1771-1778.

Magdaong, E. T., Fujii, M., Yamano, H., Licuanan, W. Y., Maypa, A., Campos, W. L., ... & Martinez, R. (2014). Long-term change in coral cover and the effectiveness of marine protected areas in the Philippines: a meta-analysis. Hydrobiologia, 733 (1), 5-17.

Mangi, S. C., & Roberts, C. M. (2006). Quantifying the environmental impacts of artisanal fishing gear on Kenya's coral reef ecosystems. Marine pollution bulletin, 52 (12), 1646-1660.

Manly, B. F., & Alberto, J. A. N. (Eds.). (2014). Introduction to ecological sampling. CRC Press.

Mantelatto, M. C., Fleury, B. G., Menegola, C., & Creed, J. C. (2013). Cost–benefit of different methods for monitoring invasive corals on tropical rocky reefs in the southwest Atlantic. Journal of experimental marine biology and ecology, 449, 129-134.

McAllister, D. E. (1991). What is the status of the world's coral reef fishes? Sea Wind 5, 14-18.

McCarthy, M. A., Moore, J. L., Morris, W. K., Parris, K. M., Garrard, G. E., Vesk, P. A., ... & Friend, T. (2013). The influence of abundance on detectability. Oikos, 122 (5), 717-726.

McClanahan, T. R., & Arthur, R. (2001). The effect of marine reserves and habitat on populations of East African coral reef fishes. Ecological Applications, 11 (2), 559-569.

McClanahan, T. R., Cinner, J. E., Maina, J., Graham, N. A. J., Daw, T. M., Stead, S. M., ... & Polunin, N. V. C. (2008). Conservation action in a changing climate. Conservation letters, 1 (2), 53-59.

McClanahan, T. R., Graham, N. A., Wilson, S. K., Letourneur, Y., & Fisher, R. (2009). Effects of fisheries closure size, age, and history of compliance on coral reef fish communities in the western Indian Ocean. Marine Ecology Progress Series, 396, 99-109.

McCorry, D. (2005). Community-Based Marine Protected Area Programme - Technical Report. Project Seahorse Foundation for Marine Conservation, Cebu City, Philippines.

McCulloch, M., Falter, J., Trotter, J., & Montagna, P. (2012). Coral resilience to ocean acidification and global warming through pH up-regulation. Nature Climate Change, 2 (8), 623.

McLeod, E., Salm, R., Green, A., & Almany, J. (2009). Designing marine protected area networks to address the impacts of climate change. Frontiers in Ecology and the Environment, 7(7), 362-370.

McManus, J. W., & Polsenberg, J. F. (2004). Coral–algal phase shifts on coral reefs: ecological and environmental aspects. Progress in Oceanography, 60 (2), 263-279.

Merino, G., Barange, M., Blanchard, J. L., Harle, J., Holmes, R., Allen, I., ... & Jennings, S. (2012). Can marine fisheries and aquaculture meet fish demand from a growing human population in a changing climate? Global Environmental Change, 22 (4), 795-806.

Meyers, L. M., DeKeyser, E. S., & Norland, J. E. (2014). Differences in spatial autocorrelation (SAc), plant species richness and diversity, and plant community composition in grazed and ungrazed grasslands along a moisture gradient, North Dakota, USA. Applied Vegetation Science, 17 (1), 53-62.

Moberg, F., & Folke, C. (1999). Ecological goods and services of coral reef ecosystems. Ecological economics, 29 (2), 215-233.

Molloy, P. P., Evanson, M., Nellas, A. C., Rist, J. L., Marcus, J. E., Koldewey, H. J., & Vincent, A. C. J. (2013). How much sampling does it take to detect trends in coral‐reef habitat using photoquadrat surveys?. Aquatic Conservation: Marine and Freshwater Ecosystems, 23 (6), 820-837.

Mora, C., Aburto-Oropeza, O., Bocos, A. A., Ayotte, P. M., Banks, S., Bauman, A. G., ... & Brooks, A. (2011). Global human footprint on the linkage between biodiversity and ecosystem functioning in reef fishes. PLoS biology, 9 (4), e1000606.

Mora, C., Andréfouët, S., Costello, M. J., Kranenburg, C., Rollo, A., Veron, J., ... & Myers, R. A. (2006). Coral reefs and the global network of marine protected areas. SCIENCE-NEW YORK THEN WASHINGTON-, 2006, 1750.

Mosquera, I., Côté, I. M., Jennings, S., & Reynolds, J. D. (2000, November). Conservation benefits of marine reserves for fish populations. In Animal Conservation forum (Vol. 3, No. 4, pp. 321-332). Cambridge University Press.

Mumby, P. J., & Harborne, A. R. (2010). Marine reserves enhance the recovery of corals on Caribbean reefs. Plos one, 5 (1), e8657.

Mumby, P. J., & Steneck, R. S. (2008). Coral reef management and conservation in light of rapidly evolving ecological paradigms. Trends in ecology & evolution, 23 (10), 555-563.

Mumby, P. J., Dahlgren, C. P., Harborne, A. R., Kappel, C. V., Micheli, F., Brumbaugh, D. R., ... & Buch, K. (2006). Fishing, trophic cascades, and the process of grazing on coral reefs. science, 311 (5757), 98-101.

Mumby, P. J., Harborne, A. R., Williams, J., Kappel, C. V., Brumbaugh, D. R., Micheli, F., ... & Blackwell, P. G. (2007). Trophic cascade facilitates coral recruitment in a marine reserve. Proceedings of the National Academy of Sciences, 104 (20), 8362-8367.

Nadon, M. O., & Stirling, G. (2006). Field and simulation analyses of visual methods for sampling coral cover. Coral Reefs, 25 (2), 177-185.

Nakajima, R., Nakayama, A., Yoshida, T., Kushairi, M. R. M., Othman, B. H. R., & Toda, T. (2010). An evaluation of photo line-intercept transect (PLIT) method for coral reef monitoring. Galaxea, Journal of Coral Reef Studies, 12 (1), 37-44.

Ohlhorst, S. L., Liddell, W. D., Taylor, R. J., & Taylor, M. J. (1988). Evaluation of reef census techniques. In Sixth Int. Coral Reef Symp (Vol. 2, p. 319).

Olds, A. D., Pitt, K. A., Maxwell, P. S., & Connolly, R. M. (2012). Synergistic effects of reserves and connectivity on ecological resilience. Journal of Applied Ecology, 49 (6), 1195-1203.

Palmer, M. W., & McGlinn, D. J. (2017). Scale detection using semivariograms and autocorrelograms. In Learning landscape ecology (pp. 65-81). Springer New York.

Pandolfi, J. M., Bradbury, R. H., Sala, E., Hughes, T. P., Bjorndal, K. A., Cooke, R. G., ... & Warner, R. R. (2003). Global trajectories of the long-term decline of coral reef ecosystems. Science, 301 (5635), 955-958.

Pandolfi, J. M., Connolly, S. R., Marshall, D. J., & Cohen, A. L. (2011). Projecting coral reef futures under global warming and ocean acidification. science, 333 (6041), 418-422.

Paulay, G. (1997). Diversity and distribution of reef organisms. In: Birkeland, C., (Ed.), Life and Death of Coral Reefs (pp. 298-345). Chapman and Hall, New York, NY.

Pauly, D., Christensen, V., Guénette, S., & Pitcher, T. J. (2002). Towards sustainability in world fisheries. Nature, 418 (6898), 689.

Peterson, C. H., & Lubchenco, J. (1997). On the value of marine ecosystems to society. In: Daily, G.C. (Ed.), Nature's Services. Societal Dependence on Natural Ecosystems (pp. 177-194). New York, NY: Island Press.

Pinheiro, J. C., & Bates, D. M. (2000). Mixed-Effects Models in S and S-plus. Statistics and computing.

Polunin, N. V. C., & Roberts, C. M. (1993). Greater biomass and value of target coral-reef fishes in two small Caribbean marine reserves. Marine ecology progress series, 167-176.

Pomeroy, R. S., Parks, J. E., & Watson, L. M. (2004). How is your MPA doing?: a guidebook of natural and social indicators for evaluating marine protected area management effectiveness. IUCN.

Preskitt, L. B., Vroom, P. S., & Smith, C. M. (2004). A rapid ecological assessment (REA) quantitative survey method for benthic algae using photoquadrats with scuba. Pacific Science, 58 (2), 201-209.

Project Seahorse Foundation for Marine Conservation. (2013). MPA LTM LIT metadata. Cebu City, Philippines.

Protected Planet. (2017). Explore the World's Marine Protected Areas. Retrieved from https://protectedplanet.net/marine

Roberts, C. M., McClean, C. J., Veron, J. E., Hawkins, J. P., Allen, G. R., McAllister, D. E., ... & Vynne, C. (2002). Marine biodiversity hotspots and conservation priorities for tropical reefs. Science, 295 (5558), 1280-1284.

Rodwell, L. D., Barbier, E. B., Roberts, C. M., & McClanahan, T. R. (2003). The importance of habitat quality for marine reserve fishery linkages. Canadian Journal of Fisheries and Aquatic Sciences, 60 (2), 171-181.

Sale, P. F., Cowen, R. K., Danilowicz, B. S., Jones, G. P., Kritzer, J. P., Lindeman, K. C., ... & Steneck, R. S. (2005). Critical science gaps impede use of no-take fishery reserves. Trends in ecology & evolution, 20 (2), 74-80.

Samoilys, M. A., Martin-Smith, K. M., Giles, B. G., Cabrera, B., Anticamara, J. A., Brunio, E. O., & Vincent, A. C. (2007). Effectiveness of five small Philippines' coral reef reserves for fish populations depends on site-specific factors, particularly enforcement history. Biological conservation, 136 (4), 584-601.

Sant, N., Chappuis, E., Rodríguez-Prieto, C., Real, M., & Ballesteros, E. (2017). Cost-benefit of three different methods for studying Mediterranean rocky benthic assemblages. Scientia Marina, 81 (1), 129-138.

Šaškov, A., Dahlgren, T. G., Rzhanov, Y., & Schläppy, M. L. (2015). Comparison of manual and semi-automatic underwater imagery analyses for monitoring of benthic hard-bottom organisms at offshore renewable energy installations. Hydrobiologia, 756 (1), 139-153.

Schaffelke, B., Carleton, J., Skuza, M., Zagorskis, I., & Furnas, M. J. (2012). Water quality in the inshore Great Barrier Reef lagoon: Implications for long-term monitoring and management. Marine Pollution Bulletin, 65 (4), 249-260.

Schleyer, M. H., & Tomalin, B. J. (2000). Damage on South African coral reefs and an assessment of their sustainable diving capacity using a fisheries approach. Bulletin of Marine Science, 67 (3), 1025-1042.

Schmeller, D. S., HENRY, P. Y., Julliard, R., Gruber, B., Clobert, J., Dziock, F., ... & Kull, T. (2009). Advantages of volunteer‐based biodiversity monitoring in Europe. Conservation biology, 23 (2), 307-316.

Selgrath, J. C. (2017). Quantifying the development of small-scale fisheries on coral reefs, and their impact on habitats (Doctoral dissertation). Retrieved from https://open.library.ubc.ca/cIRcle/collections/ubctheses/24/items/1.0354473

Selig, E. R., & Bruno, J. F. (2010). A global analysis of the effectiveness of marine protected areas in preventing coral loss. PLoS One, 5 (2), e9278.

Smith, S. V. (1978). Coral-reef area and the contributions of reefs to processes and resources of the world's oceans. Nature, 273 (5659), 225-226.

Spalding, M. D., & Grenfell, A. M. (1997). New estimates of global and regional coral reef areas. Coral Reefs, 16 (4), 225-230.

Spalding, M. D., Meliane, I., Milam, A., Fitzgerald, C., & Hale, L. Z. (2012). Aichi Target 11: Reshaping the global agenda for MPAs. Policy Brief for the Nature Conservancy.

Spalding, M., Ravilious, C., & Green, E. P. (2001). World atlas of coral reefs. Univ of California Press.

Spurgeon, J. P. (1992). The economic valuation of coral reefs. Marine Pollution Bulletin, 24 (11), 529-536.

Suarez, D. (2009). MPA LIT line intercept transect (LIT) benthic survey briefing.

The Coral Triangle Atlas. (2017). Retrieved from http://ctatlas.reefbase.org/

Toropova, C., Meliane, I., Laffoley, D., Matthews, E., & Spalding, M. (2010). Global ocean protection: present status and future possibilities. IUCN.

Trygonis, V., & Sini, M. (2012). photoQuad: a dedicated seabed image processing software, and a comparative error analysis of four photoquadrat methods. Journal of Experimental Marine Biology and Ecology, 424, 99-108.

UN. (2015). Transforming our world: the 2030 Agenda for Sustainable Development.

Underwood, A. J. (1997). Experiments in ecology: their logical design and interpretation using analysis of variance. Cambridge University Press.

UNEP-WCMC. (2006). In the Front Line: Shoreline Protection and Other Ecosystem Services from Mangroves and Coral Reefs. UNEP-WCMC, Cambridge, UK.

Van Rein, H., Schoeman, D. S., Brown, C. J., Quinn, R., & Breen, J. (2011). Development of benthic monitoring methods using photoquadrats and scuba on heterogeneous hard‐substrata: a boulder‐slope community case study. Aquatic Conservation: Marine and Freshwater

Ecosystems, 21 (7), 676-689.

Walmsley, S. F., & White, A. T. (2003). Influence of social, management and enforcement factors on the long-term ecological effects of marine sanctuaries. Environmental Conservation, 30 (4), 388-407.

Weinberg, S. (1981). A comparison of coral reef survey methods. Bijdragen tot de Dierkunde, 51 (2), 199-218.

Wilkinson, C. R., & Souter, D. N. (Eds.). (2008). Status of Caribbean coral reefs after bleaching and hurricanes in 2005 (Vol. 148). Global Coral Reef Monitoring Network.

Wilkinson, C., & Salvat, B. (2012). Coastal resource degradation in the tropics: does the tragedy of the commons apply for coral reefs, mangrove forests and seagrass beds. Marine pollution bulletin, 64 (6), 1096-1105.

Yacci, P. (2009). Feature selection of microarray data using genetic algorithms and artificial neural networks. Rochester Institute of Technology.

Yasué, M., Kaufman, L., & Vincent, A. C. J. (2010). Assessing ecological changes in and around marine reserves using community perceptions and biological surveys. Aquatic Conservation: Marine and Freshwater Ecosystems, 20 (4), 407-418.

Yasué, M., Nellas, A., & Vincent, A. C. J. (2012). Seahorses helped drive creation of marine protected areas, so what did these protected areas do for the seahorses?. Environmental Conservation, 39 (2), 183-193.

Yoshikawa, T., & Asoh, K. (2004). Entanglement of monofilament fishing lines and coral death. Biological Conservation, 117 (5), 557-560.

Zhu, X., & Zhang, J. (2009). Quartered neighbor method: a new distance method for density estimation. Frontiers of Biology in China, 4 (4), 574.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., & Smith, G. M. (2009a). Dealing with heterogeneity. In Mixed effects models and extensions in ecology with R (pp. 71-100). Springer New York.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., & Smith, G. M. (2009b). Violation of Independence – Part II. In Mixed effects models and extensions in ecology with R (pp. 161-191). Springer New York.

Appendices

Appendix A

Table A Descriptions of benthic sub-categories (modified from Project Seahorse Foundation for Marine Conservation, 2013)

Abiotic	
Rock	Hard, non-coral derived substrates.
Sand	Small particles. Return to the bottom when disturbed without increasing local turbidity.
Silt	Small particles. Remain in water column when disturbed and increase local turbidity.

Algae	
Algal assemblage	Green, interconnecting bubble algae or combination of two or more algal species.
Coral algae	Encrusting, hard calcareous algae typically violet, orange, or pinkish in color.
Halimeda	Calcareous green algae with linked individual small leaves.
Macro-algae	Fleshy algae typically green, red, or brown.
Sargassum	Brown algae which form large vertical stands often with small grape-like bubbles.
Turf algae	Filamentous algae which form fuzzy or lush turfs over reef substrates.

Hard coral	
Acropora	Horizontal table-like forms of branching coral.
Coral branching	Coral with obvious branching which typically includes secondary branching.
Coral encrusting	Coral colonies which confirm to the surface of underlying substrates.
Coral foliose	Plating coral (vertical or horizontal) which forms whirls or tiers of overlying plates.
Coral massive	Large, boulder structure with corallites which forms roundish (brain-like) shapes.
Coral mushroom	Lives unattached to substrates. Flat or dome-shaped coral consisting of one corallite.
Coral sub-massive	Smaller than massive coral and often adopts a near-branching form at the outer edge.

Dead coral	
Dead coral	Recently dead as noted by the white exposed coral skeleton.
Dead coral algae	Dead coral with colonies of turf algae over the coral skeleton.
Rubble	Fragments of dead coral with various sizes.

Octocoral	
Gorgonian	Octocoral which forms sea fans or whips with polyps of eight tentacles.
Soft coral	Encrusting or fleshy, large leather coral with polyps of eight tentacles.

Invertebrate	
Anemone	Soft bodied polyps related to coral, but absent in the skeleton.
Ascidian	Soft gelatinous globular colonies or tubes which can be brightly colored.
Bivalve mollusk	Giant clams which have zooxanthella to photosynthesize.
Hydroid	Feather like colonies which form the feathery attached structure.
Sponge	Benthic animals typically brightly colored which adopt various growth forms.
Tube worm	Worms with a head composed of feather like tentacles or gills in a circular, spiral shape.
Other invertebrate	Category of the last resort when the life form cannot be identified underwater.
Zoanthid	Its polyps resembling small, encrusting colonial anemones. Embedded into substrates.

Seagrass	
Seagrass	Green marine plants which can form monospecific or multi-species beds.

Appendix B

The list of problems in marine protected area long-term monitoring data and measures taken to each problem

Table B.1 Problems of line intercept transect (LIT) data and measures taken. See Table D.1 for locations and names of data files in the database.

	Problems resulting in data errors	Descriptions of problems or / and measures taken (*italics*)
1	All raw data from the field were compiled and saved in a few large Excel spreadsheets as "Benthic_Census_RawData1 ~ 3".	Each spreadsheet has raw data collected from multiple sites. Each site consists of data from the first survey of LIT (different for each MPA) through Wet 2010 (the last survey of LIT). *Raw data were extracted by site. They were then separated by transect and saved as new spreadsheet data files for each transect.*
2	Data were not arranged chronologically.	Data of the first five sampling surveys (Wet 1998 ~ Wet 2000) were labeled as Wet 98 ~ Wet 00 whereas data from Dry 2001 ~ Wet 2010 were labeled as 1 ~ 20 each (Table E.4). Excel does not recognise Wet 98, Dry 99, etc. as numeric codes and hence does not order data with these labels chronologically. In addition, it orders data labeled as 1 ~ 20 as 1, 10, 11, … 19, 2, 20 rather than 1, 2, 3, … 18, 19, 20. This labeling system caused data arranged in an unchronological way. *Data were arranged chronologically from Wet 1998 through Wet 2010 when extracted raw data were saved as new spreadsheets.*
3	There were a number of missing data.	*All missing data were listed by site, transect, and year and season as the data tables (Table C.1).*
4	Reasons of missing data were unknown.	*Gaps in the data tables were filled in when data were found (Table C.1). Reasons for missing data were confirmed with the biologists at Zoological Society of London - Philippines (ZSL - Philippines). Missing data were considered non-existing if data were not found in Project Seahorse at the University of British Columbia and ZSL - Philippines, and no survey was confirmed to have been conducted by ZSL.*
5	Some benthic codes were misspelled.	*Misspellings of benthic codes were extracted by Excel conditional formatting and corrected.*
6	Irregular location codes were used for one of the MPAs, Handumon.	Transects with the location code A, AS, B, and BS indicate inside MPA transects, and C, CS, D, DS outside MPA transects (see Table E.3 for the location code). However, B, BS, C, and CS were found assigned to inside, and A, AS, D, and DS to outside transects for Handumon. *Any other use of irregular location codes was confirmed with the ZSL biologists (but none was found other than Handumon).*

Table B.2 Problems of photoquadrat (PQ) data and measures taken. See Table D.2 for locations and names of data files in the database.

	Problems resulting in data errors	**Descriptions of problems or / and measures taken (*italics*)**
1	Digital images, CPCe files, and Excel spreadsheets were not sorted out completely.	Some digital images of benthic communities (raw data), CPCe files (digital images with 10 random points projected by Coral Point Count with Excel extensions: CPCe), and spreadsheets (percent benthic cover data produced from CPCe) were saved in the folders for different sites, transects, and years and seasons. *These files were sorted out by site, transect, and year and season, and saved in appropriate folders.*
2	There were a number of missing digital images, CPCe files, and spreadsheets.	*Existing data were listed by site, transect, and year and season as the data tables to identify missing data (Table C.2).*
3	Reasons of missing data (digital images, CPCe files, and spreadsheets) were unknown.	*Gaps in the data tables were filled in when data were found (Table C.2). Reasons for missing data were then confirmed with the biologists at Zoological Society of London - Philippines (ZSL - Philippines). Missing data were considered non-existing if data were not found in Project Seahorse at the University of British Columbia and ZSL - Philippines, and no survey was confirmed to have been conducted by ZSL. Missing data were considered lost if the surveys were confirmed to have been conducted by the ZSL biologists, but data were not found in Project Seahorse and ZSL - Philippines.*
4	Quadrat information on spreadsheets were not arranged numerically.	Many of the photoquadrat images were labeled as 1, 2, 3, 4, …, 23 rather than 01, 02, 03, 04, …, 23 such that quadrats were ordered as 1, 10, 11, 12, …, 2, 21, 22, 23 by Excel when images were digitized into spreadsheets through CPCe. In addition, some quadrat information was missing in many spreadsheets, which made numerically rearranging quadrats difficult. *Spreadsheet data were extracted and copy-pasted to new spreadsheets, and quadrat information was ordered numerically.*
5	Some spreadsheets were mislabeled.	*Mislabeled files (e.g. a spreadsheet labeled as Transect 1, but stored in another folder) were corrected. Data were considered mislabeled when site and transect information in the file's name in the spreadsheet did not match with that in the columns' names in the spreadsheet (each column corresponding to data for each quadrat). The file's name in the spreadsheet was considered accurate when it matched with that of the digital images and CPCe files stored in the same folder as well as information in the columns' name.*

Appendix C

Data tables showing the presence of original data files (data before cleaning) of marine protected area long-term monitoring in Danajon Bank

Table C.1 Data tables showing the presence of line intercept transect (LIT) raw data by MPA. Data availability is shown by transect for each MPA (laid out from west to east), and they are: (1) ID: Inside Deep; (2) IS: Inside Shallow; (3) OD: Outside Deep; and (4) OS: Outside Shallow. For Asinan, where there is no outside area, transects are: (1) GH D: Guard House Deep; (2) GH S: Guard House Shallow; (3) Post D: Post Deep; and (4) Post S: Post Shallow. Numbers in the tables indicate the number of transects. A grey color indicates no survey conducted. See Table D.1 "Step 1" for locations and names of the files.

Year	Season	Bilang Bilangan Transect				Batasan Transect				Pandanon Transect				Asinan Transect			
		ID	IS	OD	OS	ID	IS	OD	OS	ID	IS	OD	OS	GH D	GH S	Post D	Post S
1998	Wet	2		1						2		1		2		1	
1999	Dry	2		1		1		1		2		1		1		1	
1999	Wet	2	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2000	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2000	Wet					2	2	2	2								
2001	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2001	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2002	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2002	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2003	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2003	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2004	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2004	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2005	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2005	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2006	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2006	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2007	Dry																
2007	Wet																
2008	Dry																
2008	Wet	3	2	2	2	2	2	2	2	2	2	1	2				
2009	Dry													1	2	2	2
2009	Wet																
2010	Dry	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2
2010	Wet	3	2	2	2	2	2	2	2	2	2	1	2	1	2	2	2

Year	Season	Jandayan Norte Transect				Handumon Transect				Pinamgo Transect				Bantiguian Transect			
		ID	IS	OD	OS	ID	IS	OD	OS	ID	IS	OD	OS	ID	IS	OD	OS
1998	Wet																
1999	Dry					2		1									
1999	Wet					2	2	2	2								
2000	Dry					2	2	2	2								
2000	Wet					2	2	2	2								
2001	Dry					2	2	2	2								
2001	Wet					2	2	2	2								
2002	Dry					2	2	2	2								
2002	Wet					2	2	2	2	2	2	1	1				
2003	Dry	2	2	1	1	2	2	2	2	2	2	2	2				
2003	Wet	2	2	2	2	2	2	2	2	2	2	2	2				
2004	Dry	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2004	Wet	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2005	Dry	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2005	Wet	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2006	Dry	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2006	Wet	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2007	Dry																
2007	Wet																
2008	Dry																
2008	Wet	2	2	2	2	2	2	2	2								
2009	Dry					2	2	2	2	2	2	2	2	2	2	2	2
2009	Wet																
2010	Dry	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2010	Wet	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2

Table C.2 Data tables showing the presence of photoquadrat (PQ) data. Data availability is shown by transect (indicated as T1 ~ 10) for each MPA. Existing data are indicated as "i" (digital images), "c" (CPCe files), and "e" (Excel spreadsheets). A grey color indicates no survey conducted. See Table D.2 "Step 1" for locations and names of the files.

Table C.2.1 Data tables showing the presence of PQ data for inside and outside Bilang Bilangan MPA

Year	Season	Bilang Bilangan Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										
Year	Season	Bilang Bilangan Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.2 Data tables showing the presence of PQ data for inside and outside Batasan MPA

Year	Season	Batasan Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet										
2014	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Year	Season	Batasan Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet										
2014	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.3 Data tables showing the presence of PQ data for inside and outside Pandanon MPA

Year	Season	Pandanon Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet										
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										
Year	Season	Pandanon Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet										
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.4 Data tables showing the presence of PQ data for Guard House and Post areas of Asinsn MPA

Year	Season	Asinan Guard House Area Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet										
2008	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Wet										
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet										
2014	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Year	Season	Asinan Post Area Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet										
2008	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Wet										
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet										
2014	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.5 Data tables showing the presence of PQ data for inside and outside Jandayan Norte MPA. A yellow color indicates lost data files. Missing data were considered lost if the surveys were confirmed to have been conducted by the ZSL biologists, but data were not found in Project Seahorse at the University of British Columbia and ZSL - Philippines.

Year	Season	Jandayan Norte Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet	i	i	i	i	i	i	i	i	i	i
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										
Year	Season	Jandayan Norte Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e			i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet										
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet	i	i	i	i	i	i	i	i	i	i
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.6 Data tables showing the presence of PQ data for inside and outside Handumon MPA

Year	Season	Handumon Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet	i	i	i	i	i	i	i	i	i	i
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										
Year	Season	Handumon Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Dry										
2008	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Dry										
2009	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet	i	i	i	i	i	i	i	i	i	i
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.7 Data tables showing the presence of PQ data for inside and outside Pinamgo MPA

Year	Season	Pinamgo Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet										
2008	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Wet										
2009	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Wet										
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet	i	i	i	i	i	i	i	i	i	i
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										
Year	Season	Pinamgo Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet										
2008	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Wet										
2009	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2009	Wet										
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet	i	i	i	i	i	i	i	i	i	i
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Table C.2.8 Data tables showing the presence of PQ data for inside and outside Bantiguian MPA

Year	Season	Bantiguian Inside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet										
2008	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Wet										
2009	Dry										
2009	Wet										
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										
Year	Season	Bantiguian Outside Transect									
		T1	T2	T3	T4	T5	T6	T7	T8	T9	T10
2007	Dry										
2007	Wet										
2008	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2008	Wet										
2009	Dry										
2009	Wet										
2010	Dry	e	e	e	e	e	e	e	e	e	e
2010	Wet	e	e	e	e	e	e	e	e	e	e
2011	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2011	Wet										
2012	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2012	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2013	Wet	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2014	Dry	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e	i/e
2014	Wet										
2015	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2015	Wet										
2016	Dry	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e	i/c/e
2016	Wet										

Appendix D

Steps taken to transform data for analyses from original data, explained as file descriptions with file name examples, file name descriptions, and saving folders. All sorted data files and transformed data are saved in a newly set database "IwaoFujii_Data_Rcodes". All files in the same step are labeled and saved in the same manner. Arrows in "*saving folder*" indicate the direction to the subfolder the files are saved.

Table D.1 File descriptions with file name examples, file name descriptions, and saving folders of line intercept transect (LIT) for each step

Step 1	*File description*: LIT raw data : Each file compiling data from multiple sites and from the first to the last survey (Wet 2010) of LIT *File name example (format)*: Benthic_Census_RawData1 (Excel: .xls) *File name description*: NA *Saving folder*: IwaoFujii_Data_Rcodes -> MPA_LTM_Database -> 20_LIT_Raw_Data
Step 2	*File description*: Data extracted from LIT raw data by transect, data arranged chronologically : Each file for one transect, and all years and seasons included *File name example (format)*: bain_lit_a (Excel: .csv) *File name description*: ba (MPA code) in (inside or outside) _ lit (LIT) _ a (location code) *Saving folder*: IwaoFujii_Data_Rcodes -> LTM -> data -> batasan -> batasan_in -> bain_lit
Step 3	*File description*: Data subsampled based on the Step 2 files using R statistical software : Each file for one transect, and all years and seasons included *File name example (format)*: baina_litdf (Excel: .csv) *File name description*: ba (MPA code) in (inside or outside) a (location code) _ lit (LIT) df (dataframe) *Saving folder*: IwaoFujii_Data_Rcodes -> LTM -> csv -> df_by_site -> batasan -> batasan_in
Step 4	*File description*: Files compiling all transect data in the Step 3 files, formatted for data analyses using R : Each file for one site with all transect data *File name example (format)*: bain_litdf (Excel: .csv) *File name description*: ba (MPA code) in (inside or outside) _ lit (LIT) df (dataframe) *Saving folder*: IwaoFujii_Data_Rcodes -> LTM -> csv -> df_by_site -> batasan -> batasan_in

Table D.2 File descriptions with file name examples, file name descriptions, and saving folders of photoquadrat (PQ) for each step

Step 1.1	*File description*: Digital images of benthic communities Each file for one photoquadrat *File name example (format)*: BA_IN_T1_(01) (digital images: .jpg) *File name description*: BA (MPA code) _ IN (inside or outside) _ T1 (transect code) _ (01) (quadrat code) *Saving folder*: IwaoFujii_Data_Rcodes -> MPA_LTM_Database -> 03_Batasan -> 2007_WET_BA -> BA_IN -> BA_IN_T1
Step 1.2	*File description*: Benthic images with 10 random points projected by Coral Point Count with Excel extensions (CPCe) Each file for one photoquadrat *File name example (format)*: BA_IN_T1_(01) (CPCe: .cpce) *File name description*: BA (MPA code) _ IN (inside or outside) _ T1 (transect code) _ (01) (quadrat code) *Saving folder*: IwaoFujii_Data_Rcodes -> MPA_LTM_Database -> 03_Batasan -> 2007_WET_BA -> BA_IN -> BA_IN_T1
Step 1.3	*File description*: Files produced out of digital images through CPCe : Each file for one transect with all quadrat data *File name example (format)*: BA_IN_T1 (Excel: .xlsx) *File name description*: BA (MPA code) _ IN (inside or outside) _ T1 (transect code) *Saving folder*: IwaoFujii_Data_Rcodes -> MPA_LTM_Database -> 03_Batasan -> 2007_WET_BA -> BA_IN -> BA_IN_T1
Step 2	*File description*: Files compiling all transect data in the Step 1.3 files : Each file for one year with all transect data *File name example (format)*: ba_in_2007wet (Excel: .xlsx) *File name description*: ba (MPA code) _ in (inside or outside) _ 2007wet (year and season) *Saving folder*: IwaoFujii_Data_Rcodes -> LTM -> data -> batasan -> batasan_in -> bain_pq
Step 3	*File description*: Files compiling all year data in the Step 2 files, formatted for data analyses using R : Each file for one site with all year data *File name example (format)*: bain_pqdf (Excel: .csv) *File name description*: ba (MPA code) in (inside or outside) _ pq (photoquadrat) df (dataframe) *Saving folder*: IwaoFujii_Data_Rcodes -> LTM -> csv -> df_by_site -> batasan -> batasan_in

Appendix E

Transect code for line intercept transect (LIT) and quadrat code for photoquadrat (PQ)

Table E.1 The structure of the LIT transect code: B for benthic - site / location - year season

Transect number	Transect code
1	B-BA A-16
2	B-BA AS-16
3	B-BA B-16
4	B-BA BS-16
5	B-BA C-16
6	B-BA CS-16
7	B-BA D-16
8	B-BA DS-16

Table E.2 MPA code for the LIT transects (same for the PQ survey)

MPA code	Marine protected area
BB	Bilang Bilangan
BA	Batasan
PA	Pandanon
BV	Asinan (Buenavista)
JN	Jandayan Norte
HA	Handumon
PI	Pinamgo
SA	Bantiguian (Saguise)

Table E.3 Location code for the LIT transects

Location code	Location
A	Inside 1 / Deep
AS	Inside 1 / Shallow
B	Inside 2 / Deep
BS	Inside 2 / Shallow
C	Outside 1 / Deep
CS	Outside 1 / Shallow
D	Outside 2 / Deep
DS	Outside 2 / Shallow

Table E.4 Year season code for the LIT transects

Year season code	Year and season
98W	Wet 1998
99D	Dry 1999
99W	Wet 1999
00D	Dry 2000
00W	Wet 2000
1	Dry 2001
2	Wet 2001
3	Dry 2002
4	Wet 2002
5	Dry 2003
6	Wet 2003
7	Dry 2004
8	Wet 2004
9	Dry 2005
10	Wet 2005
11	Dry 2006
12	Wet 2006
16	Wet 2008
17	Dry 2009
18	Wet 2009
19	Dry 2010
20	Wet 2010

Table E.5 Quadrat code for the PQ survey

Example code
Code: BA_IN_T1_(01)
Structure: MPA code _ location code (IN or OUT) _ transect code (T1~10) _ quadrat code (01~23)

Appendix F

Results of the preliminary analyses conducted for the analysis of differences in mean percent coral cover between line intercept transect (LIT) and photoquadrat (PQ). The five subsampling treatments were tested, each of which consists of 22 mean values. Each mean represents mean percent coral cover of data in one sampling survey at one of eight MPAs between Wet 2008 and Wet 2010. The five treatments were LIT data before and after subsampling (indicated as LIT line and LIT point data, respectively), and PQ data with all quadrats (sample n = 230), and 80 and 40 randomly selected quadrats (indicated as PQ 230, PQ 80, and PQ 40 data, respectively).

Table F.1 Results of the Shapiro-Wilk test to examine the normal distribution for each of the five subsampling treatments: (1) LIT line data; (2) LIT point data; (3) PQ 230 data; (4) PQ 80 data; and (5) PQ 40 data. A Bonferroni corrected p-value ($\alpha = 0.01$) was used as a critical value.

Subsampling treatment	W value	p-value
LIT line data	0.901	0.031
LIT point data	0.935	0.158
PQ 230 data	0.916	0.063
PQ 80 data	0.933	0.141
PQ 40 data	0.966	0.618

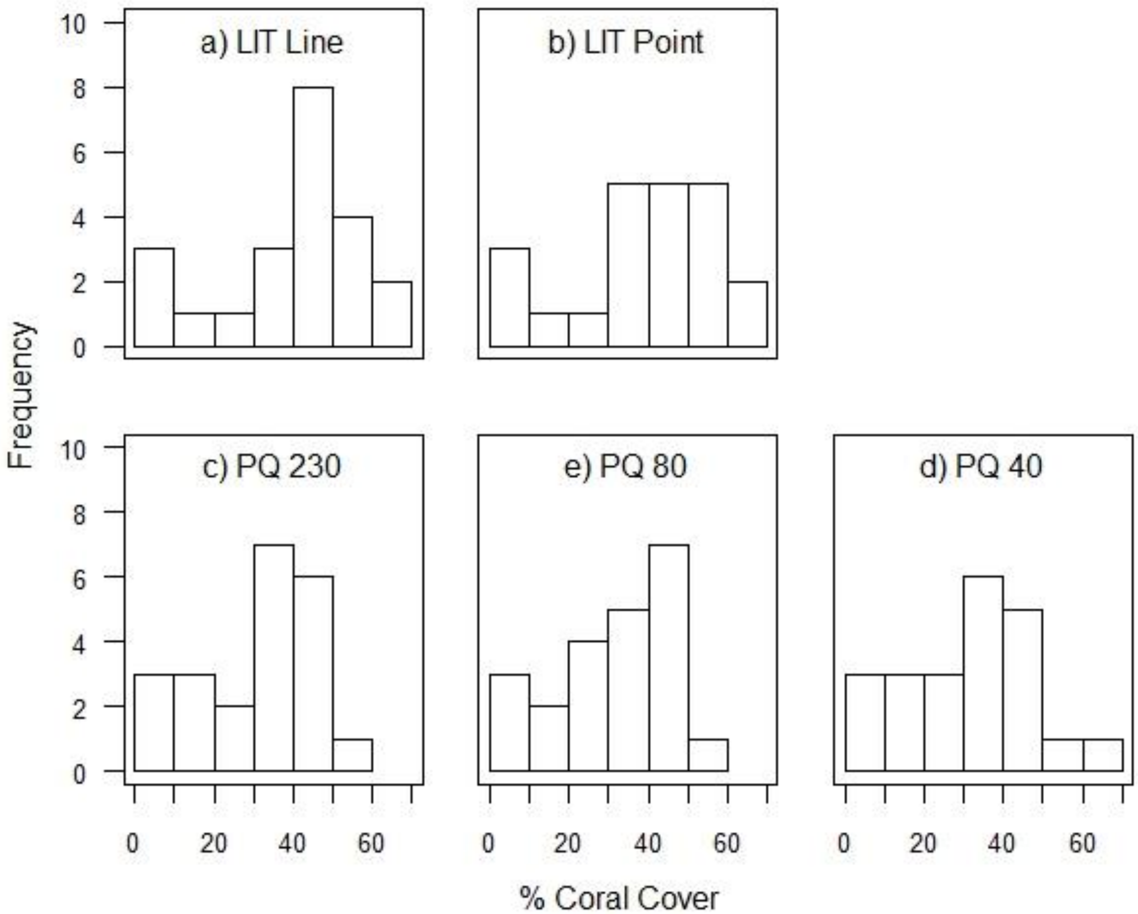

Fig. F.1 Distributions of data for each of the five subsampling treatments: (a) LIT line data; (b) LIT point data; (c) PQ 230 data; (d) PQ 80 data; and (e) PQ 40 data. Each data point represents a mean of data collected in one sampling survey at one of eight MPAs between Wet 2008 and Wet 2010.

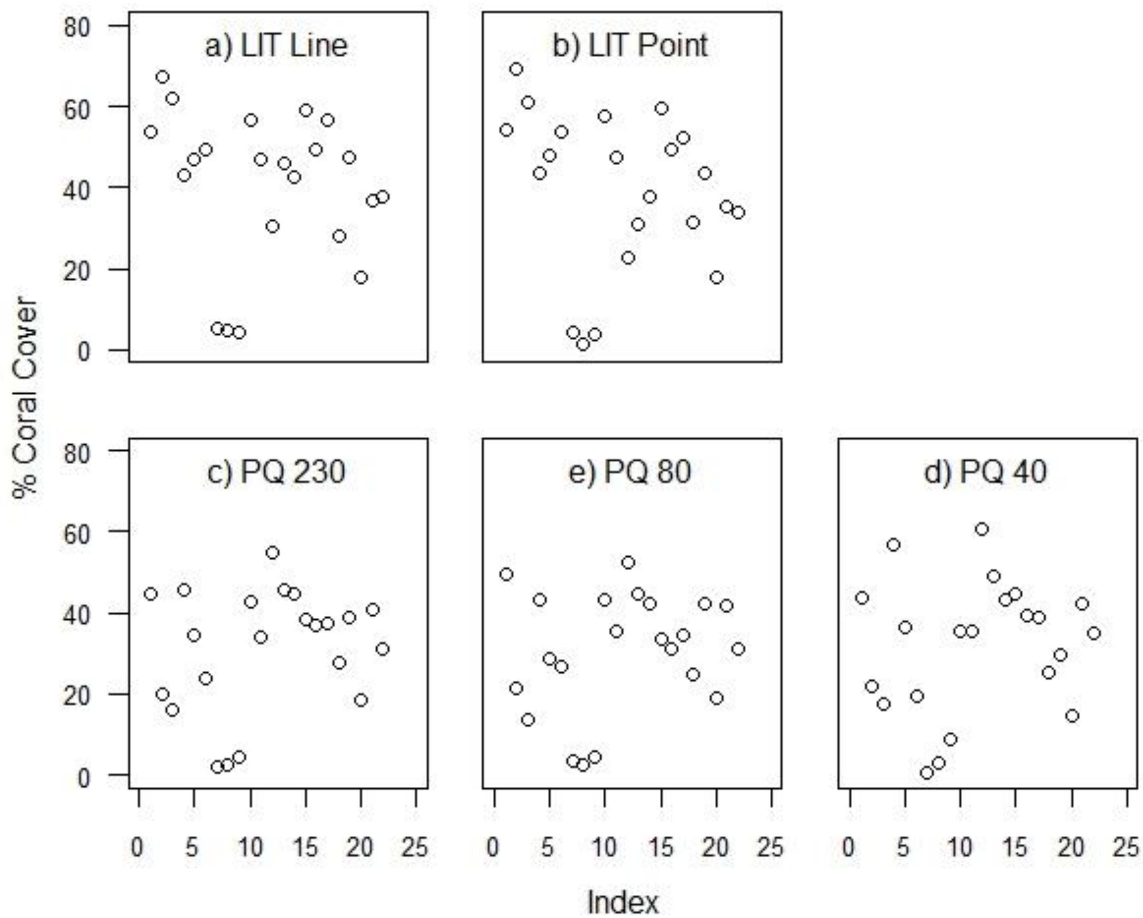

Fig. F.2 The variance of data for each of the five subsampling treatments: (a) LIT line data; (b) LIT point data; (c) PQ 230 data; (d) PQ 80 data; and (e) PQ 40 data. Each data point represents a mean of data collected in one sampling survey at one of eight MPAs between Wet 2008 and Wet 2010.

Table F.2 Results of the F-test to examine differences in data variance between LIT and PQ for six pairs of subsampling treatments: (1) LIT line and PQ 230 data; (2) LIT line and PQ 80 data; (3) LIT line and PQ 40 data; (4) LIT point and PQ 230 data; (5) LIT point and PQ 80 data; and (6) LIT point and PQ 40 data. A Bonferroni corrected p-value ($\alpha = 0.0083$) was used as a critical value.

Pair examined	F value	p-value
LIT line vs PQ 230	1.521	0.344
LIT line vs PQ 80	1.568	0.310
LIT line vs PQ 40	1.294	0.560
LIT point vs PQ 230	1.656	0.256
LIT point vs PQ 80	1.708	0.228
LIT point vs PQ 40	1.409	0.438

Appendix G

Table G Results of the comparison in mean percent coral cover between line intercept transect (LIT) and photoquadrat (PQ) data (Wet 2008 ~ Wet 2010) with the nested structure for six pairs of subsampling treatments: (1) LIT line and PQ 230 data; (2) LIT line and PQ 80 data; (3) LIT line and PQ 40 data; (4) LIT point and PQ 230 data; (5) LIT point and PQ 80 data; and (6) LIT point and PQ 40 data. LIT line and LIT point data indicate LIT data before and after subsampling, respectively. PQ 230, PQ 80, and PQ 40 data indicate PQ data with all quadrats (sample n = 230), and 80 and 40 randomly selected quadrats, respectively. Significant results are indicated in bold, and a Bonferroni corrected p-value (α = 0.0083) was used as a critical value. SE indicates the standard error, and DF indicates the degree of freedom.

	Estimate	SE	DF	t-value	p-value
Intercept	**40.8644**	**5.2976**	**21**	**7.7138**	**0.0000**
LIT line vs PQ 230	**-9.3610**	**3.0791**	**21**	**3.0402**	**0.0062**
Intercept	**40.9107**	**5.2142**	**21**	**7.8461**	**0.0000**
LIT line vs PQ 80	**-10.0677**	**3.1749**	**21**	**3.1710**	**0.0046**
Intercept	40.8266	5.4283	21	7.5211	0.0000
LIT line vs PQ 40	-8.6063	3.2557	21	-2.6435	0.0152
Intercept	39.3603	5.3520	21	7.3543	0.0000
LIT point vs PQ 230	-7.8440	3.3387	21	2.3494	0.0287
Intercept	39.4057	5.2765	21	7.4681	0.0000
LIT point vs PQ 80	-8.5507	3.4143	21	2.5044	0.0206
Intercept	39.3222	5.4664	21	7.1934	0.0000
LIT point vs PQ 40	-7.0893	3.5262	21	2.0105	0.0574

Appendix H

Results of the Wilcoxon signed-rank test, linear mixed effects models, and linear models to examine mean percent coral cover and temporal patterns of percent coral cover estimated by the line intercept transect (LIT) and phtoquadrat (PQ) methods. LIT line and LIT point data indicate LIT data before and after subsampling, respectively. PQ 230, PQ 80, and PQ 40 data indicate PQ data with all quadrats (sample n = 230), 80 randomly selected quadrats, and 40 randomly selected quadrats, respectively. Furthermore, LIT line + PQ 230 data and LIT point + PQ 40 data here indicate LIT line and PQ 230 combined data, and LIT point and PQ 230 combined data, respectively.

Table H.1 Results of the Wilcoxon signed-rank test to examine differences between LIT and PQ data in mean percent coral cover, and differences between median values of two subsampling treatments (tested in pair) for six pairs of treatments: (1) LIT line and PQ 230 data; (2) LIT line and PQ 80 data; (3) LIT line and PQ 40 data; (4) LIT point and PQ 230 data; (5) LIT point and PQ 80 data; and (6) LIT point and PQ 40 data. The median values are the median of 22 mean values at each of the five treatments (LIT line, LIT point, PQ 230, PQ 80, and PQ 40 data). Each mean value represents a mean of data collected in one sampling survey at one of eight MPAs between Wet 2008 and Wet 2010. Significant results are indicated in bold, and a Bonferroni corrected p-value ($\alpha = 0.0083$) was used as a critical value.

Pair examined	V value	p-value	Difference
LIT line vs PQ 230	**208**	**0.0066**	**10.6105**
LIT line vs PQ 80	**219**	**0.0017**	**14.1265**
LIT line vs PQ 40	195	0.0251	11.106
LIT point vs PQ 230	189	0.0425	7.7855
LIT point vs PQ 80	188	0.0462	11.3015
LIT point vs PQ 40	181	0.0794	8.2810

Table H.2 Temporal patterns of percent coral cover (Wet 2008 ~ Wet 2010) estimated using the following subsampling treatments: (1) LIT line data; (2) LIT point data; (3) PQ 230 data; (4) PQ 40 data; (5) LIT line + PQ 230 data; and (6) LIT point + PQ 40 data. The linear mixed effects models and linear models were based on all observations rather than mean percent coral cover of each sampling survey. Significant results are indicated in bold (using a Bonferroni corrected p-value = 0.0083). SE indicates the standard error, and DF indicates the degree of freedom.

Table H.2.1 Temporal patterns of percent coral cover estimated for all MPAs combined based on the linear mixed effects models

	Estimate	SE	DF	t-value	p-value
LIT line					
Intercept	-3514.079	5687.324	35	-0.618	0.541
Year	1.769	2.830	35	0.625	0.536
LIT point					
Intercept	-2512.714	3022.305	871	-0.831	0.406
Year	1.270	1.504	871	0.844	0.399
PQ 230					
Intercept	**11786.886**	**1022.145**	**4699**	**11.532**	**< 0.001**
Year	**-5.849**	**0.509**	**4699**	**-11.501**	**< 0.001**
PQ 40					
Intercept	**15006.895**	**2518.765**	**802**	**5.958**	**< 0.001**
Year	**-7.451**	**1.253**	**802**	**-5.945**	**< 0.001**
LIT line + PQ 230					
Intercept	**11649.914**	**1015.790**	**4743**	**11.469**	**< 0.001**
Year	**-5.781**	**0.505**	**4743**	**-11.438**	**< 0.001**
LIT point + PQ 40					
Intercept	**6078.230**	**2061.679**	**1682**	**2.948**	**0.003**
Year	**-3.006**	**1.026**	**1682**	**-2.931**	**0.003**

Table H.2.2 Temporal patterns of percent coral cover estimated for Bilang Bilangan based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	-10543.773	9506.898	-1.109	0.330	0.047
Year	5.277	4.731	1.115	0.327	
LIT point					
Intercept	-10064.212	7605.278	-1.323	0.188	0.006
Year	5.038	3.784	1.331	0.186	
PQ 230					
Intercept	**29786.397**	**2286.703**	**13.030**	**< 0.001**	**0.198**
Year	**-14.808**	**1.138**	**-13.010**	**< 0.001**	
PQ 40					
Intercept	**27128.720**	**6335.559**	**4.282**	**< 0.001**	**0.128**
Year	**-13.485**	**3.153**	**-4.278**	**< 0.001**	
LIT line + PQ 230					
Intercept	**29434.860**	**2290.350**	**12.850**	**< 0.001**	**0.192**
Year	**-14.630**	**1.140**	**-12.840**	**< 0.001**	
LIT point + PQ 40					
Intercept	8292.500	5701.014	1.455	0.147	0.005
Year	-4.104	2.837	-1.447	0.149	

Table H.2.3 Temporal patterns of percent coral cover estimated for Batasan based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	-5785.473	10096.783	-0.573	0.597	-0.154
Year	2.902	5.024	0.578	0.594	
LIT point					
Intercept	-8917.788	8049.649	-1.108	0.270	0.002
Year	4.462	4.005	1.114	0.268	
PQ 230					
Intercept	**19926.706**	**2489.233**	**8.005**	**< 0.001**	**0.089**
Year	**-9.898**	**1.239**	**-7.991**	**< 0.001**	
PQ 40					
Intercept	**35171.797**	**5809.353**	**6.054**	**< 0.001**	**0.238**
Year	**-17.483**	**2.891**	**-6.048**	**< 0.001**	
LIT line + PQ 230					
Intercept	**19684.886**	**2471.506**	**7.965**	**< 0.001**	**0.087**
Year	**-9.778**	**1.230**	**-7.951**	**< 0.001**	
LIT point + PQ 40					
Intercept	12637.782	5258.505	2.403	0.017	0.020
Year	-6.267	2.617	-2.395	0.017	

Table H.2.4 Temporal patterns of percent coral cover estimated for Pandanon based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	715.696	3080.664	0.232	0.828	-0.234
Year	-0.354	1.533	-0.231	0.829	
LIT point					
Intercept	1239.635	1826.381	0.679	0.499	-0.005
Year	-0.615	0.909	-0.677	0.500	
PQ 230					
Intercept	-1684.713	855.680	-1.969	0.049	0.004
Year	0.840	0.426	1.972	0.049	
PQ 40					
Intercept	**-6768.387**	**2361.315**	**-2.866**	**0.005**	**0.059**
Year	**3.370**	**1.175**	**2.868**	**0.005**	
LIT line + PQ 230					
Intercept	-1664.167	848.671	-1.961	0.050	0.004
Year	0.830	0.422	1.964	0.049	
LIT point + PQ 40					
Intercept	-2708.525	1507.176	-1.797	0.074	0.009
Year	1.349	0.750	1.799	0.073	

Table H.2.5 Temporal patterns of percent coral cover estimated for Asinan based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	38548.050	25870.010	1.490	0.275	0.288
Year	-19.150	12.870	-1.488	0.275	
LIT point					
Intercept	42267.800	30958.300	1.365	0.176	0.011
Year	-21.000	15.400	-1.364	0.177	
PQ 230					
Intercept	33799.034	12862.245	2.628	0.009	0.021
Year	-16.794	6.398	-2.625	0.009	
PQ 40					
Intercept	396.261	34180.092	0.012	0.991	-0.023
Year	-0.180	17.003	-0.011	0.992	
LIT line + PQ 230					
Intercept	**34052.508**	**12699.688**	**2.681**	**0.008**	**0.022**
Year	**-16.920**	**6.317**	**-2.678**	**0.008**	
LIT point + PQ 40					
Intercept	31222.230	23869.300	1.308	0.193	0.006
Year	-15.510	11.870	-1.306	0.194	

Table H.2.6 Temporal patterns of percent coral cover estimated for Jandayan Norte based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	-14035.796	9579.420	-1.465	0.217	0.188
Year	7.004	4.767	1.469	0.216	
LIT point					
Intercept	-14191.923	6746.371	-2.104	0.038	0.028
Year	7.077	3.357	2.108	0.037	
PQ 230					
Intercept	**10588.812**	**2860.037**	**3.702**	**< 0.001**	**0.019**
Year	**-5.245**	**1.423**	**-3.685**	**< 0.001**	
PQ 40					
Intercept	17069.298	7355.251	2.321	0.022	0.039
Year	-8.468	3.660	-2.314	0.023	
LIT line + PQ 230					
Intercept	**10353.186**	**2837.113**	**3.649**	**< 0.001**	**0.018**
Year	**-5.128**	**1.412**	**-3.632**	**< 0.001**	
LIT point + PQ 40					
Intercept	248.648	5309.595	0.047	0.963	-0.004
Year	-0.104	2.642	-0.039	0.969	

Table H.2.7 Temporal patterns of percent coral cover estimated for Handumon based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	4909.135	21871.831	0.224	0.833	-0.235
Year	-2.415	10.883	-0.222	0.835	
LIT point					
Intercept	8633.481	7489.477	1.153	0.251	0.003
Year	-4.269	3.727	-1.146	0.254	
PQ 230					
Intercept	954.720	2708.309	0.353	0.725	-0.001
Year	-0.456	1.348	-0.339	0.735	
PQ 40					
Intercept	6054.058	6408.062	0.945	0.347	-0.001
Year	-2.992	3.189	-0.938	0.350	
LIT line + PQ 230					
Intercept	987.980	2693.199	0.367	0.714	-0.001
Year	-0.473	1.340	-0.353	0.724	
LIT point + PQ 40					
Intercept	7417.565	5011.997	1.480	0.140	0.005
Year	-3.667	2.494	-1.471	0.143	

Table H.2.8 Temporal patterns of percent coral cover estimated for Pinamgo based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	6304.543	34658.282	0.182	0.865	-0.240
Year	-3.121	17.244	-0.181	0.865	
LIT point					
Intercept	12090.000	11212.981	1.078	0.283	0.001
Year	-6.000	5.579	-1.075	0.284	
PQ 230					
Intercept	7493.297	3471.625	2.158	0.031	0.005
Year	-3.714	1.727	-2.150	0.032	
PQ 40					
Intercept	10926.207	7411.432	1.474	0.143	0.010
Year	-5.425	3.688	-1.471	0.144	
LIT line + PQ 230					
Intercept	7482.963	3450.412	2.169	0.030	0.005
Year	-3.709	1.717	-2.160	0.031	
LIT point + PQ 40					
Intercept	11508.103	6742.376	1.707	0.089	0.008
Year	-5.712	3.355	-1.703	0.090	

Table H.2.9 Temporal patterns of percent coral cover estimated for Bantiguian based on the linear models

	Estimate	SE	t-value	p-value	Adjusted R^2
LIT line					
Intercept	-4787.480	53626.030	-0.089	0.937	-0.494
Year	2.400	26.680	0.090	0.937	
LIT point					
Intercept	6065.250	34509.920	0.176	0.861	-0.012
Year	-3.000	17.170	-0.175	0.862	
PQ 230					
Intercept	**40200.637**	**11133.212**	**3.611**	**< 0.001**	**0.029**
Year	**-19.980**	**5.538**	**-3.608**	**< 0.001**	
PQ 40					
Intercept	28044.370	28813.510	0.973	0.334	-0.001
Year	-13.930	14.330	-0.972	0.335	
LIT line + PQ 230					
Intercept	**39750.221**	**11031.846**	**3.603**	**< 0.001**	**0.028**
Year	**-19.756**	**5.488**	**-3.600**	**< 0.001**	
LIT point + PQ 40					
Intercept	17346.595	22758.128	0.762	0.447	-0.003
Year	-8.611	11.321	-0.761	0.448	

Appendix I

Percent coral cover (Wet 2008 ~ Wet 2010) plotted using the following subsampling treatments: (1) LIT line data; (2) LIT point data; (3) PQ 230 data; (4) PQ 40 data; (5) LIT line + PQ 230 data; and (6) LIT point + PQ 40 data. SE indicates the standard error.

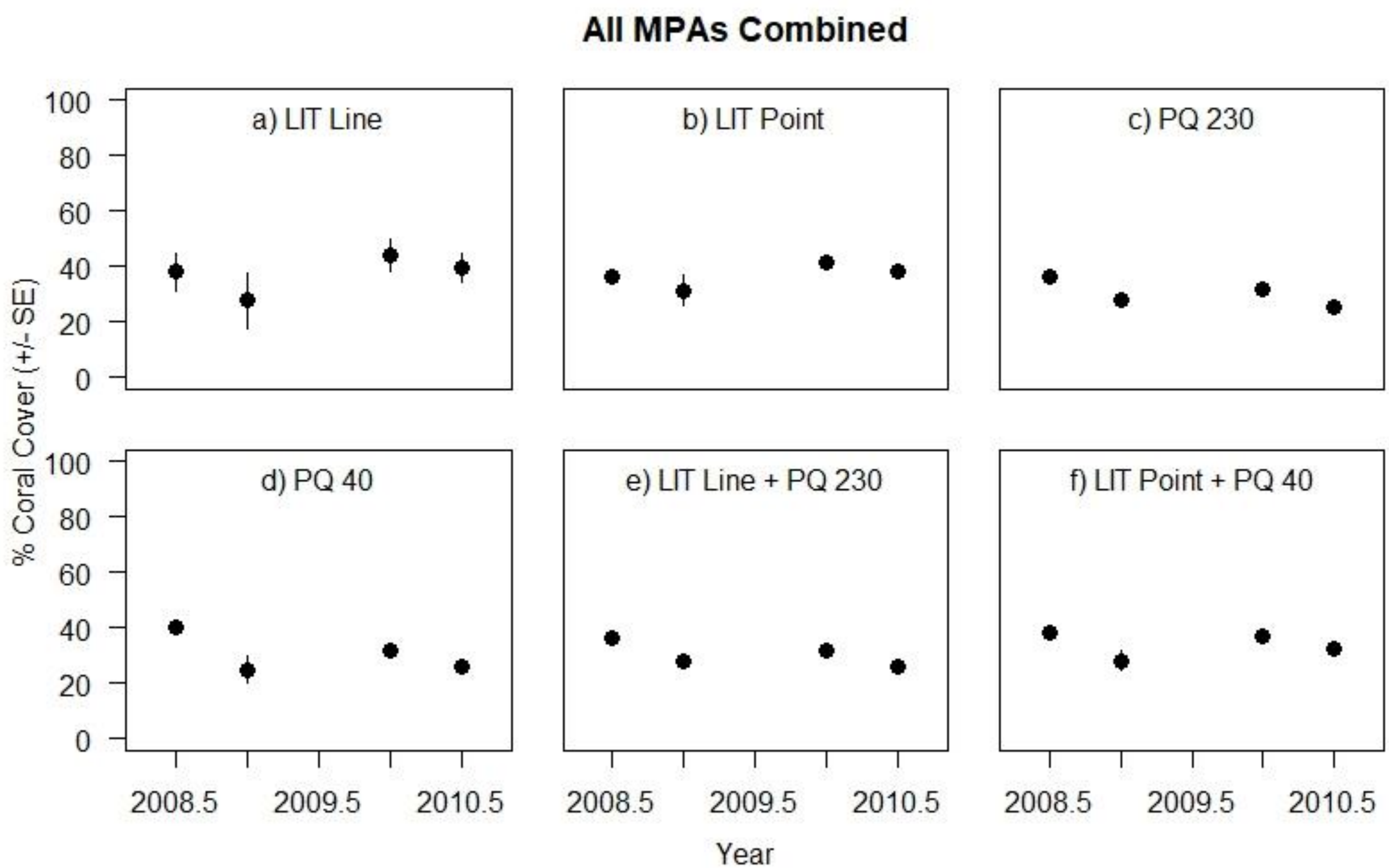

Fig. I.1 Percent coral cover during the method transition period across MPAs (data of all MPAs combined)

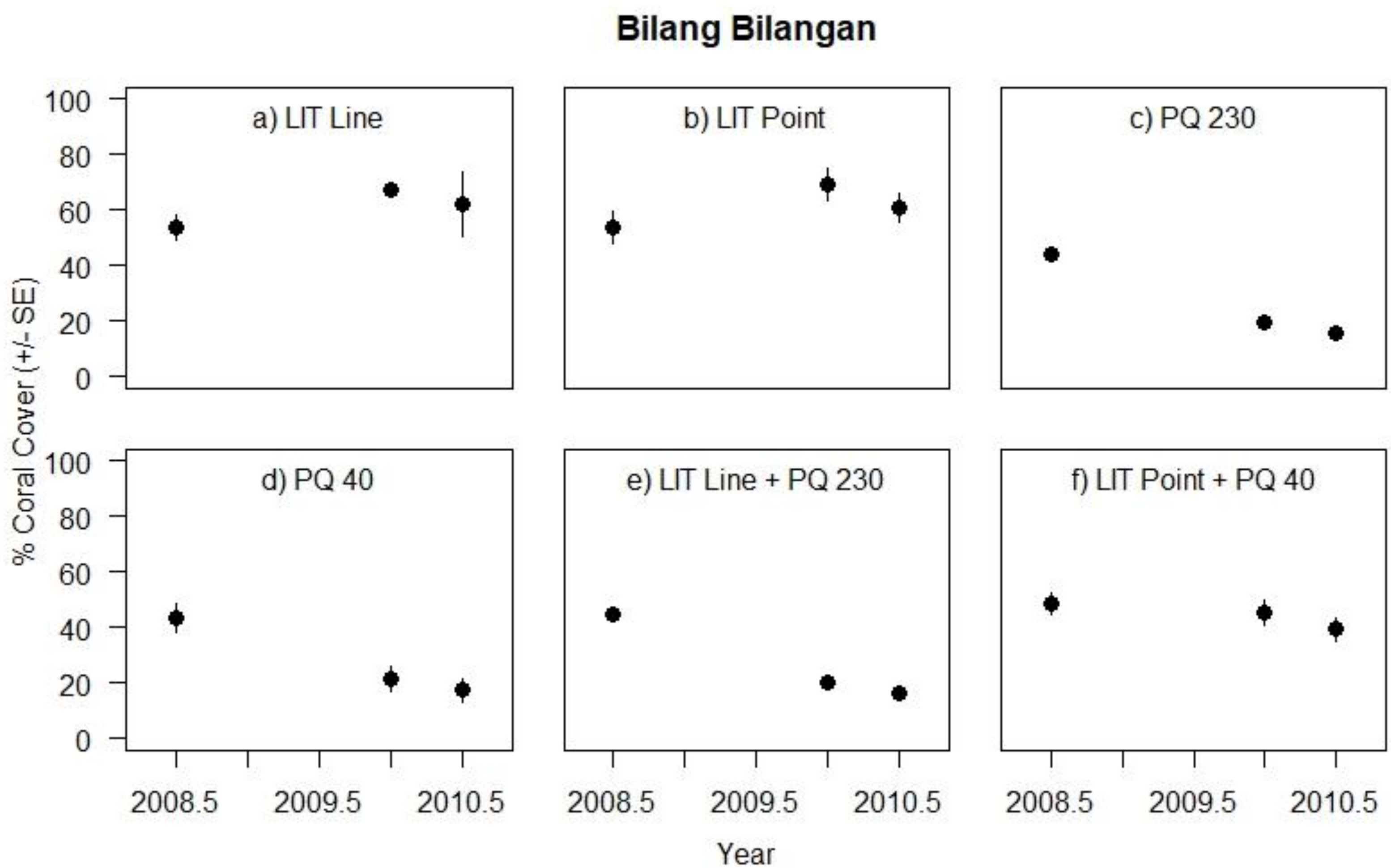

Fig. I.2 Percent coral cover during the method transition period in Bilang Bilangan

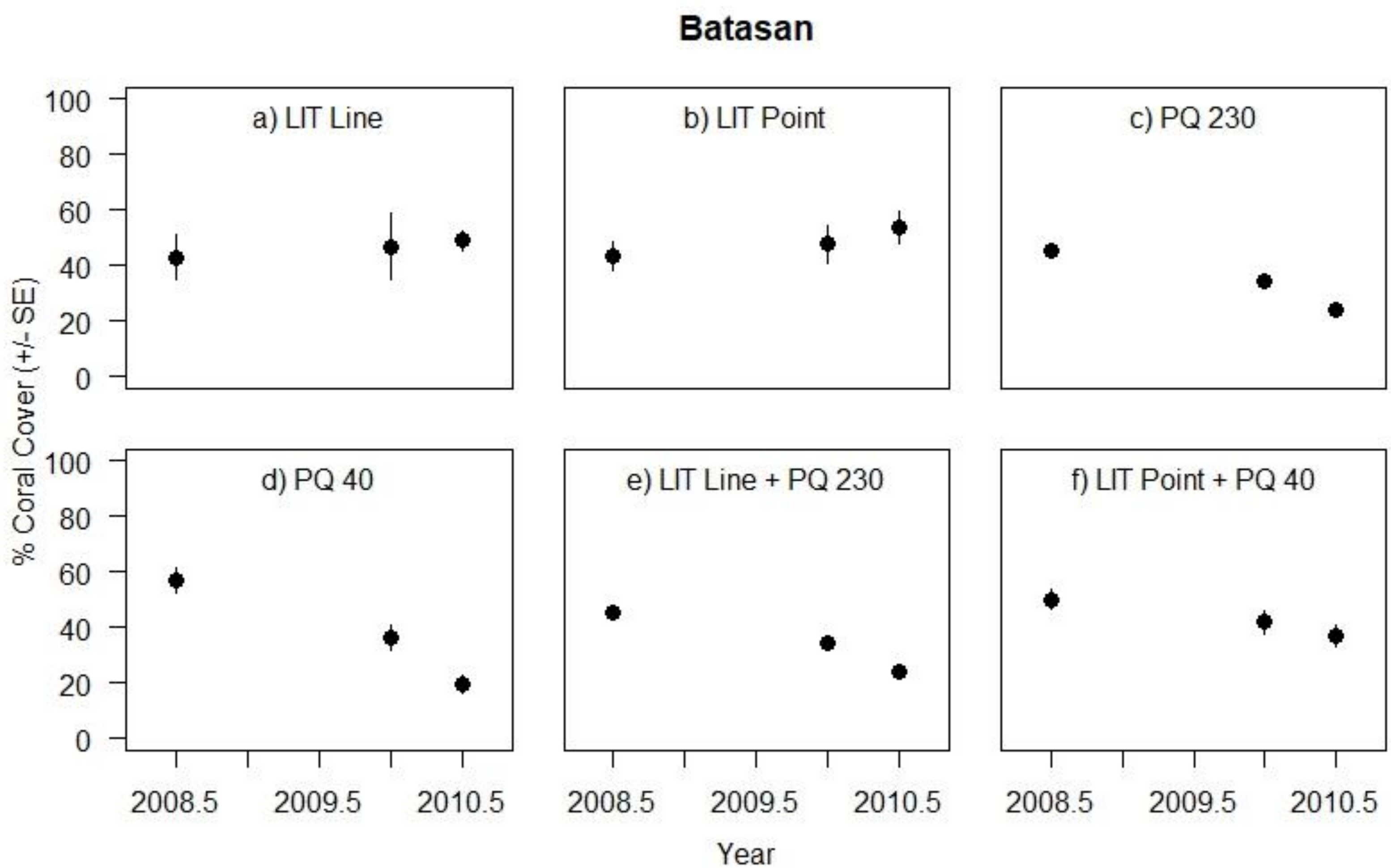

Fig. I.3 Percent coral cover during the method transition period in Batasan

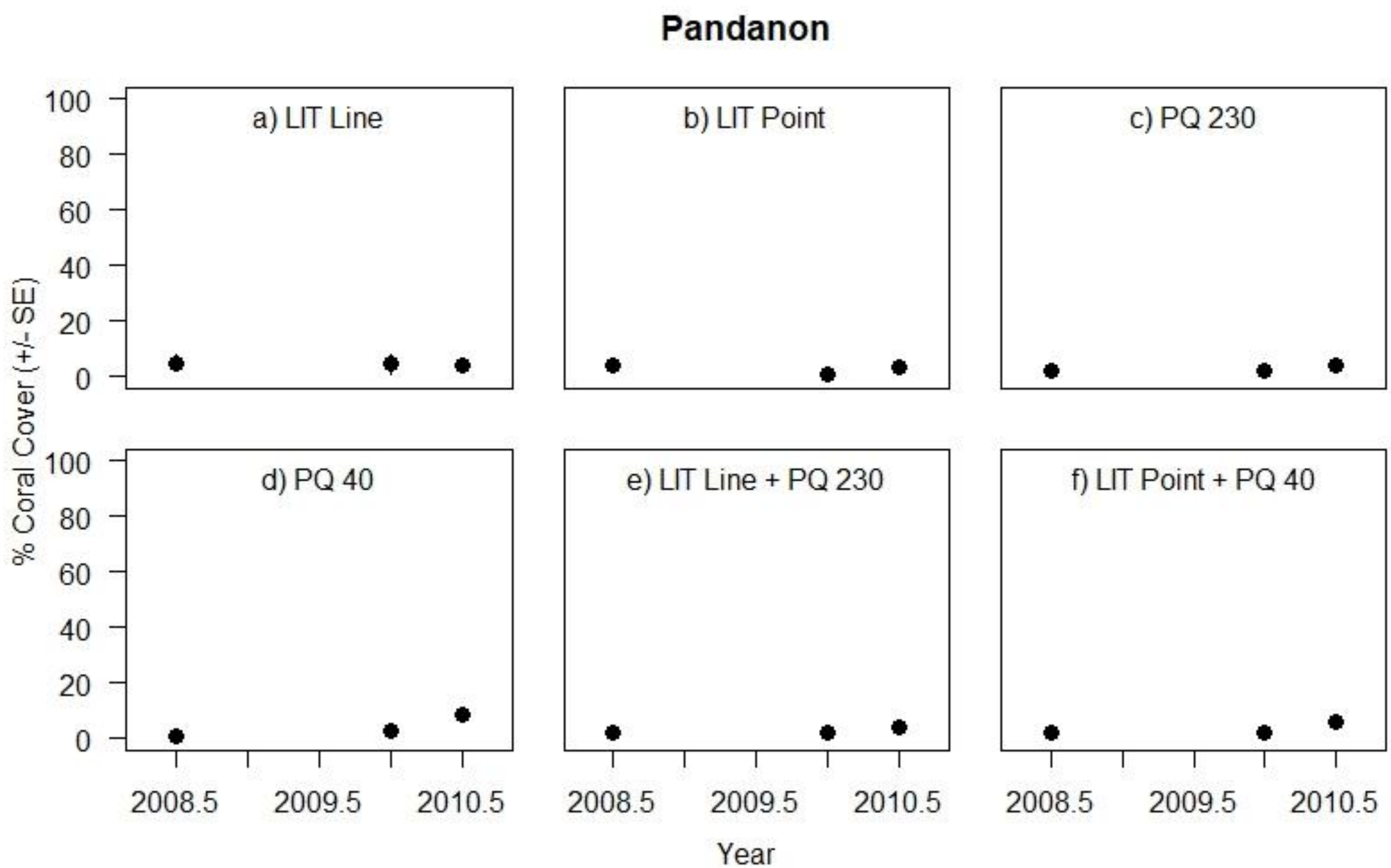

Fig. I.4 Percent coral cover during the method transition period in Pandanon

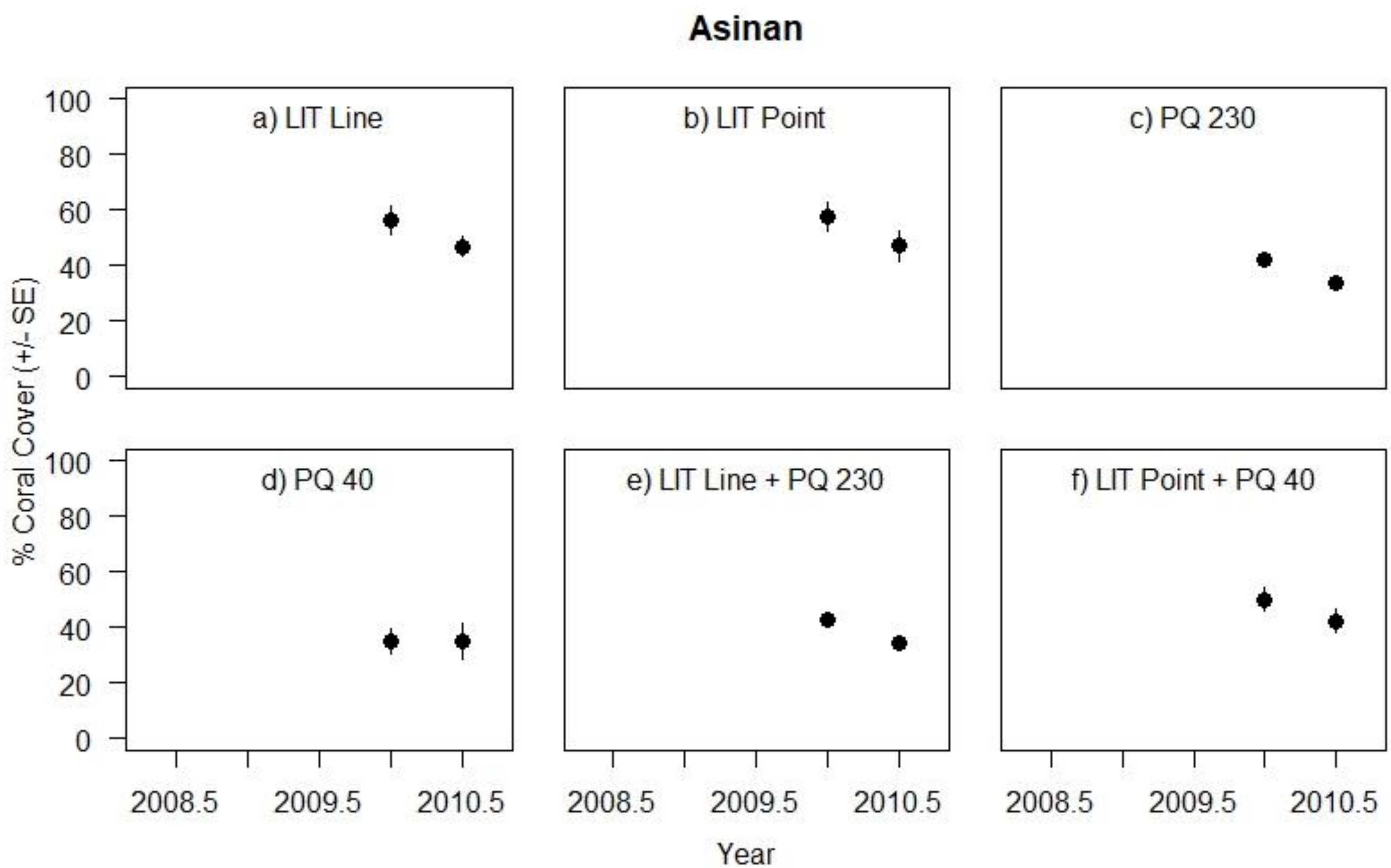

Fig. I.5 Percent coral cover during the method transition period in Asinan

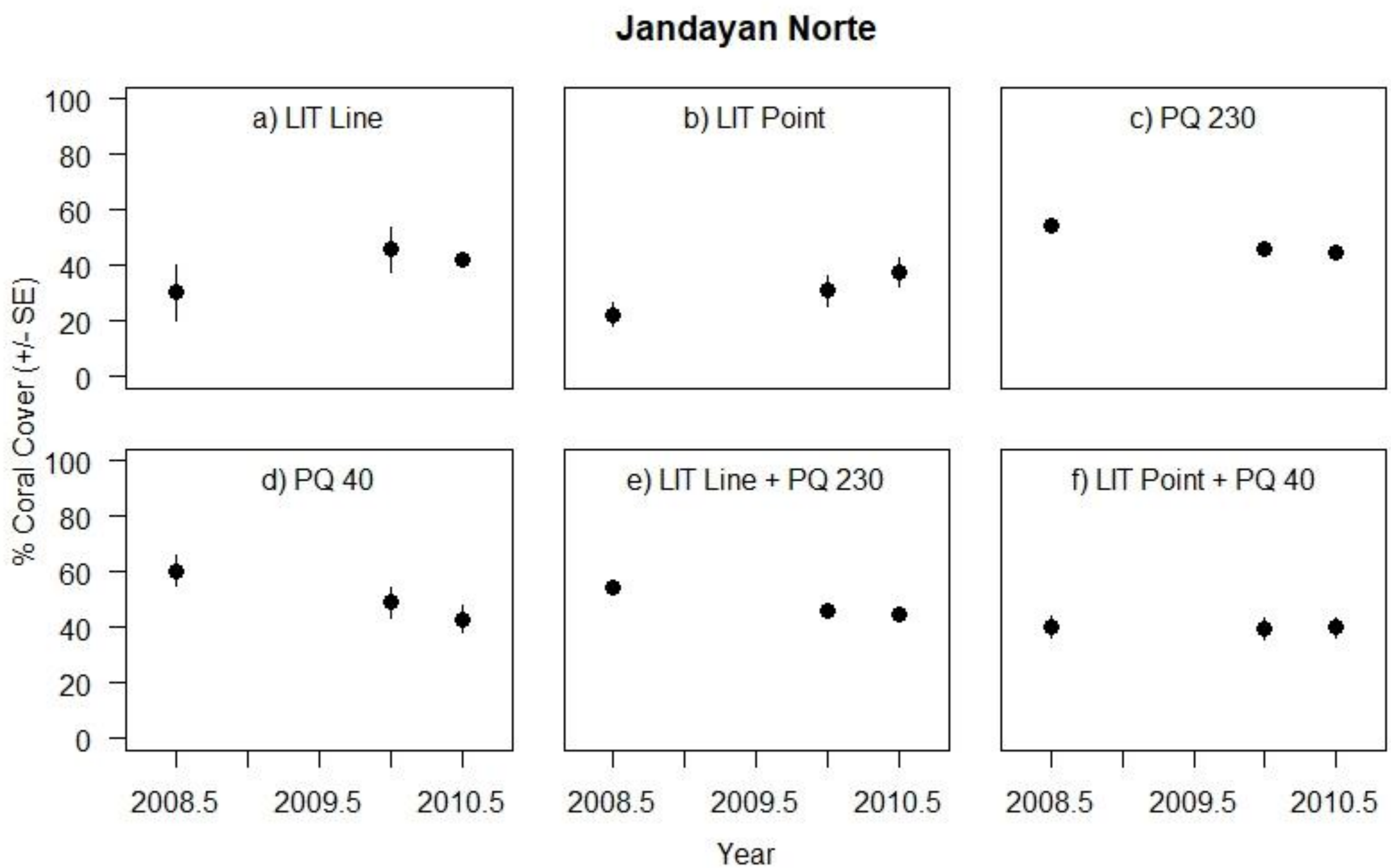

Fig. I.6 Percent coral cover during the method transition period in Jandayan Norte

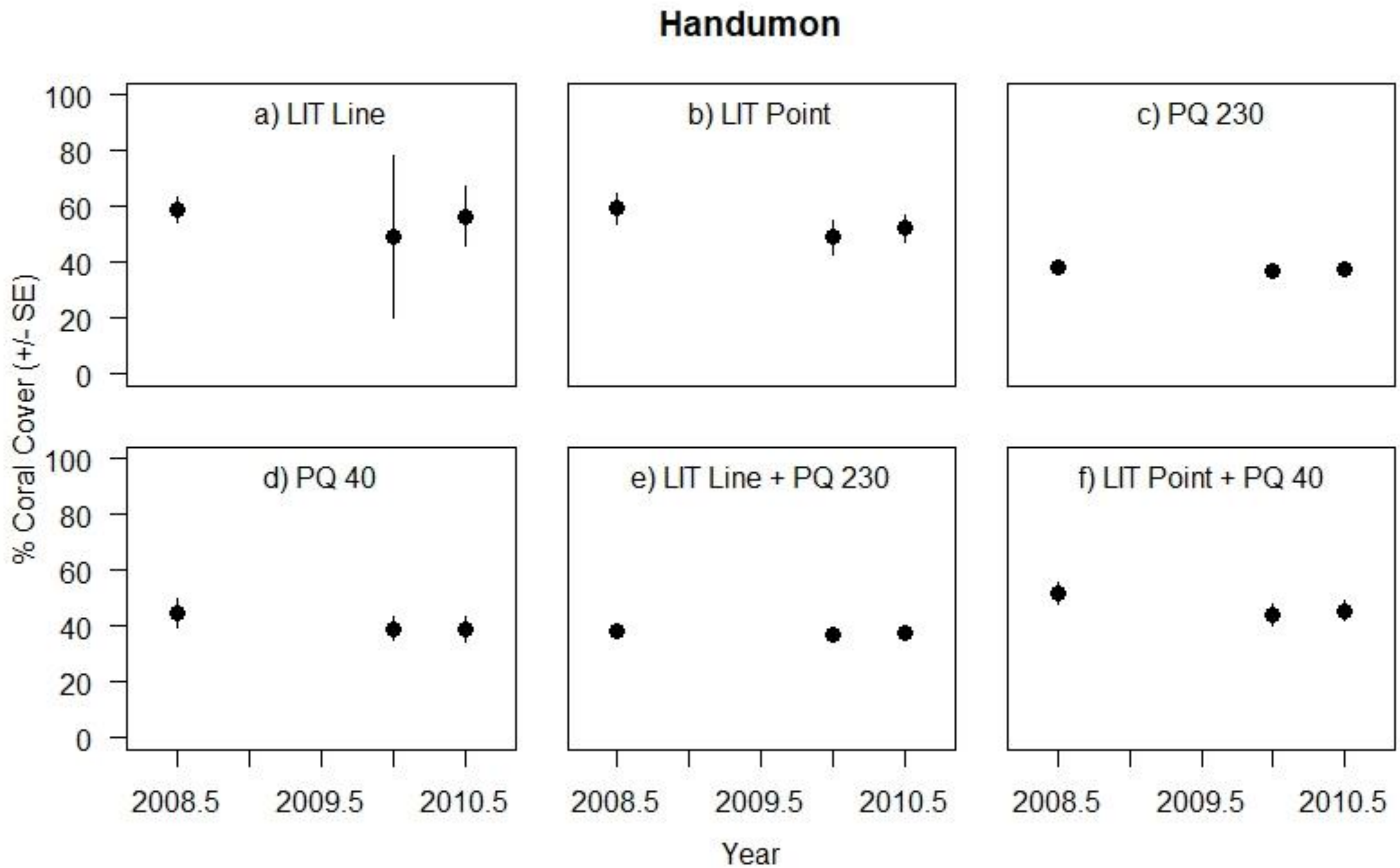

Fig. I.7 Percent coral cover during the method transition period in Handumon

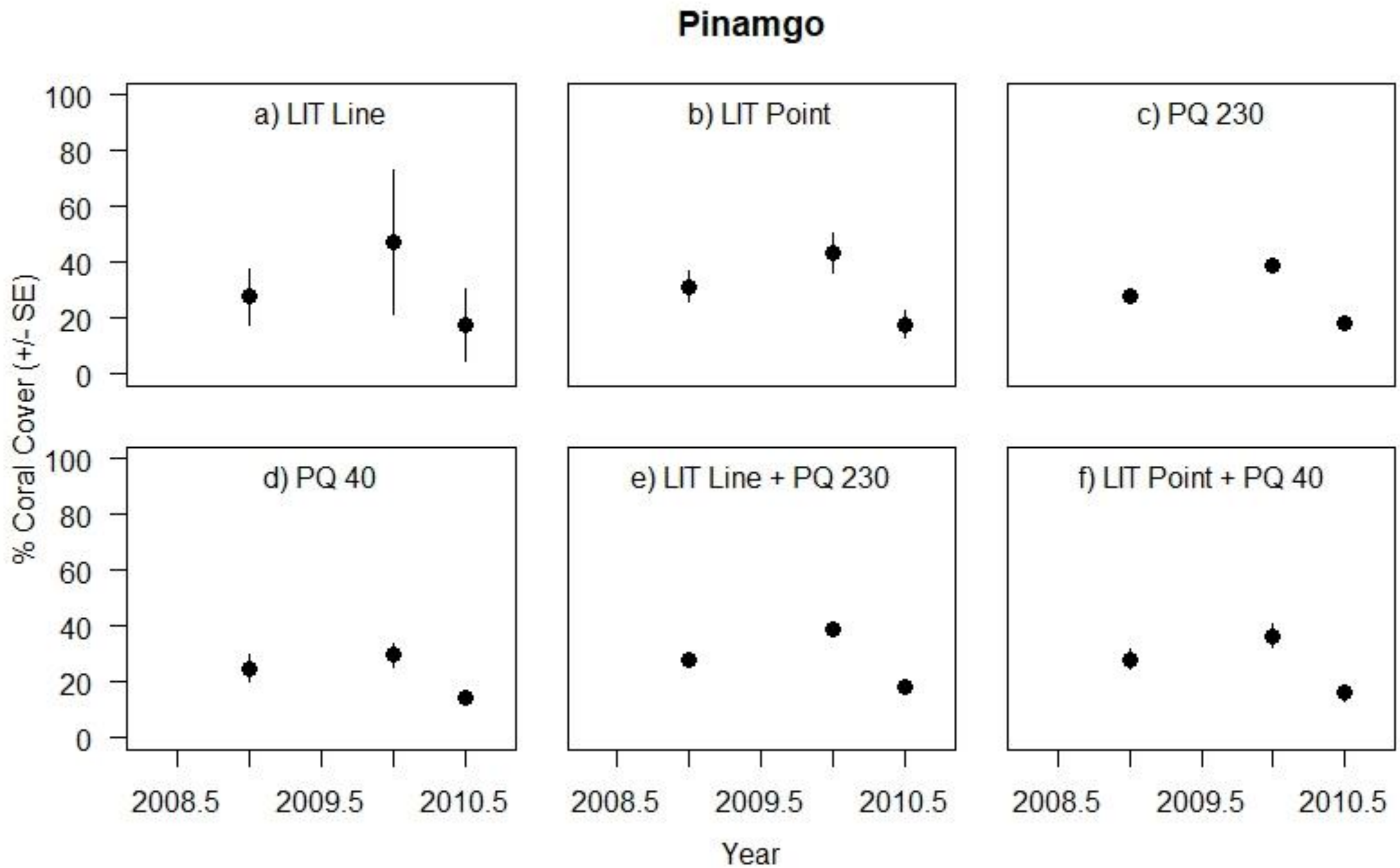

Fig. I.8 Percent coral cover during the method transition period in Pinamgo

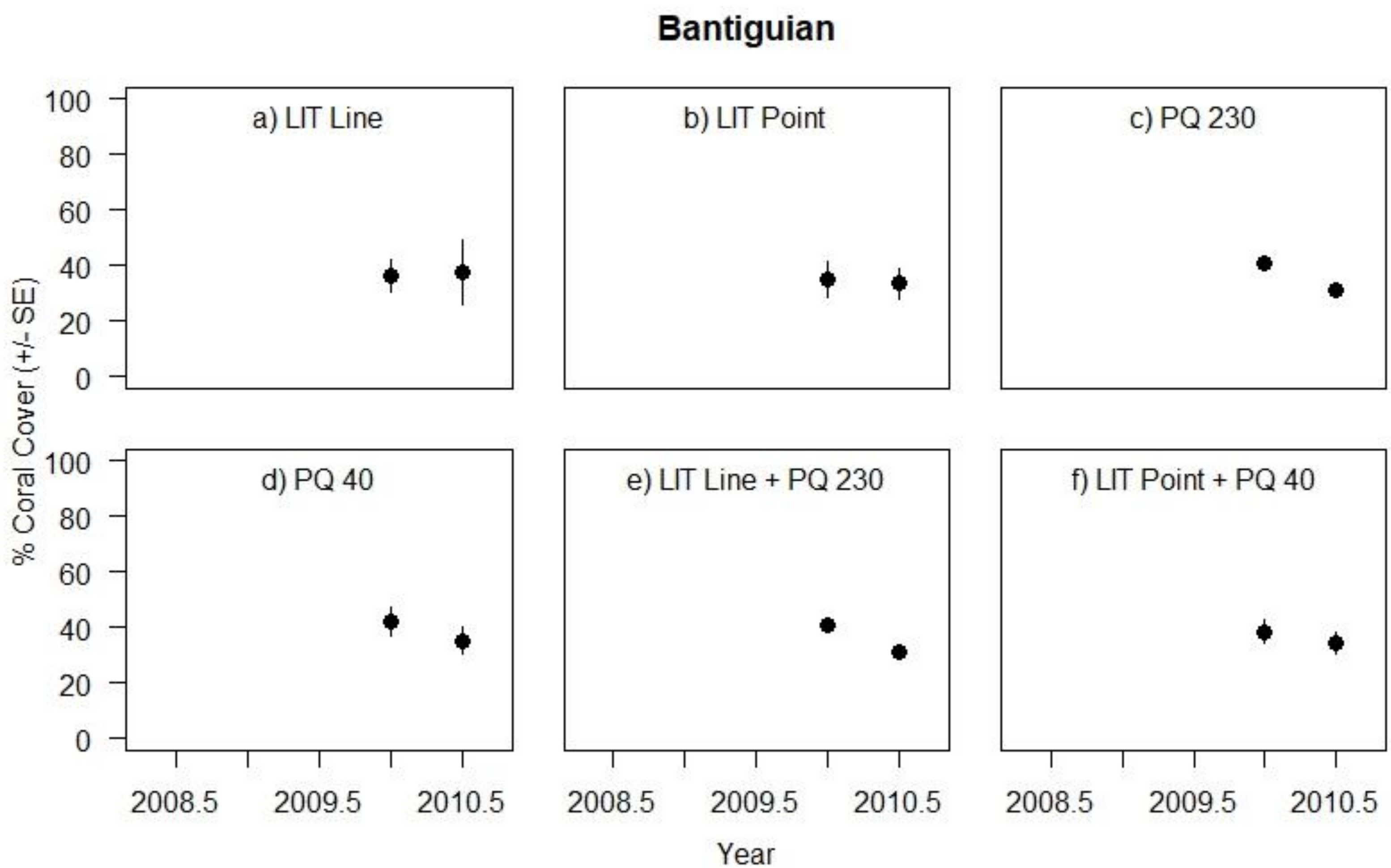

Fig. I.9 Percent coral cover during the method transition period in Bantiguian