QUANTIFYING THE DEVELOPMENT OF SMALL-SCALE FISHERIES ON CORAL REEFS, AND THEIR IMPACT ON HABITATS

by

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Abstract

Growing human populations place multiple pressures on social-ecological systems, including coastal oceans. However, the effects of long-term and/or overlapping stressors remain poorly understood, particularly over large spatial scales. My dissertation evaluated how pressures from fishing and co-occurring stressors correspond to current ecological conditions in the Danajon Bank, a coral reef ecosystem in the central Philippines. I used long-term local ecological knowledge (LEK) to map fishing practices (1950-2010) and high spatial resolution satellite imagery to map coastal habitats. This innovative suite of methods enabled me to examine patterns over broader spatial scales and longer time periods than those usually assessed. I met five primary objectives: (1) quantify the spatio-temporal dynamics of fishing effort and gear use; (2) examine the influence of fisheries governance; (3) map the spatial distribution of benthic (seafloor) habitats; (4) model the spatial distribution of living corals in relation to co-occurring stressors and biophysical conditions; and (5) explore the conservation implications of these relationships.

While individuals' fishing practices were fairly consistent over time, this small-scale fishery has changed dramatically. First, total fishing effort (days per year fished by all fishers) accelerated between 1960 and 2010 because of rapid growth in the number of fishers. Aggregate fishing effort increased almost 2.5-fold and spatially-explicit fishing effort increased over 20-fold. Second, the areal extent of fishing grounds expanded greatly. Third, use of fishing gears changed over time. Diversity of fishing gears increased, as did fishing effort with destructive, active, and non-selective gears. Considering the timing of these changes, I found a lasting influence of fishing policies, and small improvements in the sustainability of fishing gears following

implementation of co-management. Finally, I found that the probability that an area supported living corals was affected by fishing through both long- & short-term mechanisms, and I documented strong coral-landscape relationships. My research demonstrates that to strengthen ocean conservation, it will be essential to reduce the frequency and intensity of stressors, remove some areas from exploitation, foster resilience traits of ecosystems, gather data to better understand systems, and strengthen the institutions that can support these endeavors.

Lay Summary

Fishing provides an essential source of protein for half a billion people, but many fisheries damage ocean ecosystems including coral reefs. I demonstrated that small-scale fisheries in the Philippines shifted from benign to damaging through several mechanisms and had long-term effects on corals. Fishing effort increased 2.5-fold overall, and over 20-fold in some locations. Increases in total fishing effort were driven by a growing number of fishers, rather than by changes in individual fishers' behaviour. I documented a growing diversity and spatial overlap of fishing gears, with a growing amount of fishing effort dependent upon destructive, non-selective, and illegal fishing gears. Living corals were less prevalent in places that had experienced heavy fishing during the past 30 years, particularly where coral reefs were close to large villages. In contrast, corals were more likely to be present in deeper locations and in areas that were protected from fishing.

Preface

This thesis is comprised of four scientific papers of which I am (or will be) the lead author. I set project scope in collaboration with my first advisor, Dr. Amanda Vincent, with further influence from Dr. Sarah Gergel, after she became my co-advisor. I was primarily responsible for conceptualization, experimental design, collecting information (with help from a team of research assistants), data management, data analysis, interpretation of results, and manuscript preparation. My co-advisors, Dr. Amanda Vincent and Dr. Sarah Gergel were instrumental in helping me improve and refine the questions for each chapter. Additionally they assisted with interview design, provided feedback on data analyses and interpretation, and engaged heavily with the writing of the chapters. Dr. Amanda Vincent facilitated my research in the Philippines, and connected me with the Project Seahorse Foundation team (now ZSL Philippines), which acted as my local partner. For Chapter 4, Dr. Chris Roelfsema created the decision rules to classify the satellite images, provided feedback on analysis, and edited the manuscript. Dr. Roelfsema suggested the addition of the cost analysis. For Chapter 5, Dr. Gergel provided key insights about resilience.

I raised the bulk of the funding for fieldwork, and received additional support from Dr. Vincent and Dr. Gergel. Dr. Heather Koldewey (of ZSL, UK) raised a portion of the funding for Chapter 4. Dr. Phillip Molloy provided invaluable help in developing my understanding of how to manage and analyze time-series of data. Carl Walters helped me identify how to estimate trends in spatial fishing effort.

Two publications have resulted from this research:

- Chapter 3: **Selgrath, J.C.**, Gergel, S.E., and Vincent, A.C.J. (2017) "Incorporating spatial dynamics greatly increases estimates of long-term fishing effort: A participatory mapping approach." ICES Journal of Marine Science.
- Chapter 4: **Selgrath, J.C.**, Roelfsema, C., Gergel, S.E., and Vincent, A.C.J. (2016) Mapping for Coral Reef Conservation: Comparing the Value of Participatory and Remote Sensing Approaches. Ecosphere 7(5): e01325.

One chapter has been accepted to a journal with major revisions:

- Chapter 2: **Selgrath, J.C.**, Gergel, S.E., and Vincent, A.C.J. "Shifting Gears: Diversification, intensification and effort increases of small-scale fisheries" (Accepted with major revisions) PLOS One.

Chapter 5 will be submitted to a leading scientific journal as a co-authored paper with my advisors S.E. Gergel and A.C.J. Vincent.

All fieldwork in this dissertation was approved by UBC's animal care committee (A07-0076-R004) and UBC's Human Behavioural Research Ethics Board (H07-00577) and (H06-80484).

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D – Simpson's index of diversity

$cumulative\ SpE_{Fti}$ – Cumulative number of fishing days by all fishers from study communities in year (t) in grid cell (i)

e – Fishing effort, here days fished in one year

$\bar{e}_{ft}$ – Mean of individual effort (days) (e) that respondents (f) fished during one year (t)

$\bar{e}_{F,t}$ – mean individual effort (days) that all respondents (F) fished in one year (t)

e_{fti} – Spatially explicit annual fishing effort for each respondent (f) in year (t) in grid cell (i)

E_{ft} – Cumulative number of days (E) that respondents (f) reported fishing in the study area in year (t)

E_t – Total fishing effort (days) of all fishers from participating communities (F) during one year (t)

f – Individual fisher; respondent

F – Total number of fishers from participating communities

F_t – Total number of fishers in one year (t)

g – Unique fishing gear

G – Richness, here measured as number of fishing gears, also referred to as Diversity

i – Grid cell

n_t – Total number of gears of a particular type of gear used by all fishers in a year (t)

N_t – Total number of any type of gear used by all fishers in year (t)

$relative\ SpE_{fti}$ – Proportional distribution of fishing effort by all respondents (f) in year (t) in grid cell (i)

SpE_{fti} – Cumulative effort reported by all respondents (f) in year (t) in grid cell (i)

List of Abbreviations

BFAR – Bureau of fisheries and aquatic resources, Philippines

CBD – Convention on Biological Diversity

CITES – Convention on International Trade in Endangered Species of Wild Fauna and Flora

ha – Hectare

IUU – Illegal, unregulated, and unreported fishing

km – Kilometer

K-S - Kolmogorov-Smirnov test

LEK – Local environmental knowledge

MIR – Mid-infrared light

MPA – Marine protected area

MSY – Maximum sustainable yield

NIR – Near-infrared light

NGO – Non-governmental organization

OBIA – Object-based image analysis

PIT – Point intercept transect

RA – Research assistant

RA – Republic Act of the Philippines

RS – Remote sensing

SSF – Small-scale fisheries

sq – square

UBC – University of British Columbia

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I offer my immense gratitude to my two supervisors, Amanda J. Vincent and Sarah E. Gergel, who have provided me with whole-hearted support and distilled wisdom. Under your guidance I have learned to see the challenges facing ecosystems with a pragmatic, yet analytical lens. I am honored to have been your student. Thank you to my committee members – Villy Christensen and Nora Angeles. You have helped me grow as a scholar. I am incredibly grateful for the support and flexibility provided by UBC. Without a supportive university, I would not have been able to heal from the dual challenges of a near-fatal accident and training as a scientist.

I received support from BBC Wildlife Fund, Cordula and Gunter Paetzold Fellowship, Explorers Club, Fulbright Commission/PAEF, Guylian Chocolates Belgium, International Rice Research Institute, Planet Action/Digital Globe, Point Defiance Zoo and Aquarium, Project Aware, Project Seahorse Foundation (now ZSL Philippines), Rick Hansen Foundation, SeaLifeBase, Shedd Aquarium, and several sources at UBC, including Zoology and the Biodiversity Centre.

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I would not be here without my colleagues in Project Seahorse and the Landscape Ecology Lab. Danika Kleiber, Sarah Foster, Phil Molloy, and Iain Caldwell you have been friends and mentors throughout my time at UBC. Thank you to those who supported me in the Philippines: Jean Marcus, Sian Morgan, Jonathan Anticamara, James Hehre, Marivic Pajaro, Andrew Nelson, Malin Pinsky. Thanks to others who read drafts, contributed intellectually, and helped me tackle R and GIS: Lindsay Aylesworth, Kerrie O'Donnell, Stephanie Tomschea, Ian Eddy, Tanya Galliger, Kevin Yang, Kristen Dales, Kate Kirby, Ira Sutherland, Xiaofeng Ruan, Kyle Gillespie, Tíng-Chūn Kūo, Julia Lawson, Natalie Ban, Tse-Lynn Loh, Tanvi Vaidyanathan, Laura Tremblay-Boyer, Andy Knough, Kristine Stump, Michelle Paddack, and Emily Darling. Thank you to Project Seahorse including: Gina Bestbier, Scott Finestone, Heather Koldewey, and Tyler Steim. Thanks to Chuck Knapp and the Shedd Aquarium for giving me a Chicago home. Thanks to my medical team, for putting me back together. To Heidi Ravenel, Laura Humphrey, Amanda Brown, Joyce Chou, Andrea Ahrens, Tom Porteus, Emily Pearlman, Gretchen Carrigan, Maya Salas, Sarah Klain, Melissa Flagg, and Scott Kelley, thanks for being stars of inspiration. Thank you John Christiani, Sarah Keller, and Sarah Ravensbergen for your brilliance and time.

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shared your passion for education, whether through formal schooling or life's adventures.

Chapter 1: Introduction

1.1 Rationale

To sharpen the effectiveness of conservation, it's valuable to understand how human activities develop, and in turn influence ecological conditions. Fisheries are a key area in which to evaluate these feedbacks. People derive great benefits from fishing, while also exerting heavy pressures on the ocean. Catches of marine life comprise a significant source of protein and essential nutrients for approximately 1.4 billion people worldwide (FAO 2016). Yet, the fish and invertebrates captured by fisheries are also the last group of wildlife that constitutes a major part of the human diet. Under heavy fishing pressure, marine ecosystems are facing consequences similar to those faced by over-hunted terrestrial animals. These effects include depleted populations – particularly of high trophic level species, reduced size of individuals, and reduced biodiversity (Pauly et al. 1998, Bennett et al. 2002, Nañola et al. 2011). Additionally, fisheries impact the ocean by causing substantial damage to marine seafloor habitats (Watling and Norse 1998). In the ocean, these changes – combined with other global threats (e.g. ocean warming and ocean acidification) – have reduced the functioning of marine ecosystems and threaten the multitude of benefits that people derive from oceans (Moberg and Folke 1999, Worm et al. 2006, Klain and Chan 2012).

My dissertation deploys a conservation perspective in probing spatial and temporal variations in human pressures, and associated ecological conditions. My thesis is organized into three components (Figure 1.1) and I consider conservation implications of: (a) the spatio-temporal dynamics of fishing impacts, (b) the spatial distribution of benthic habitats, and (c) the spatial distribution of living corals in relation to stressors, including fishing impacts and biophysical conditions. I found fisheries to be a fascinating area to explore these relationships. Fisheries are characterized by the tension between the critical importance of feeding people and the equally important need to minimize fisheries impacts on marine wildlife and marine environments.

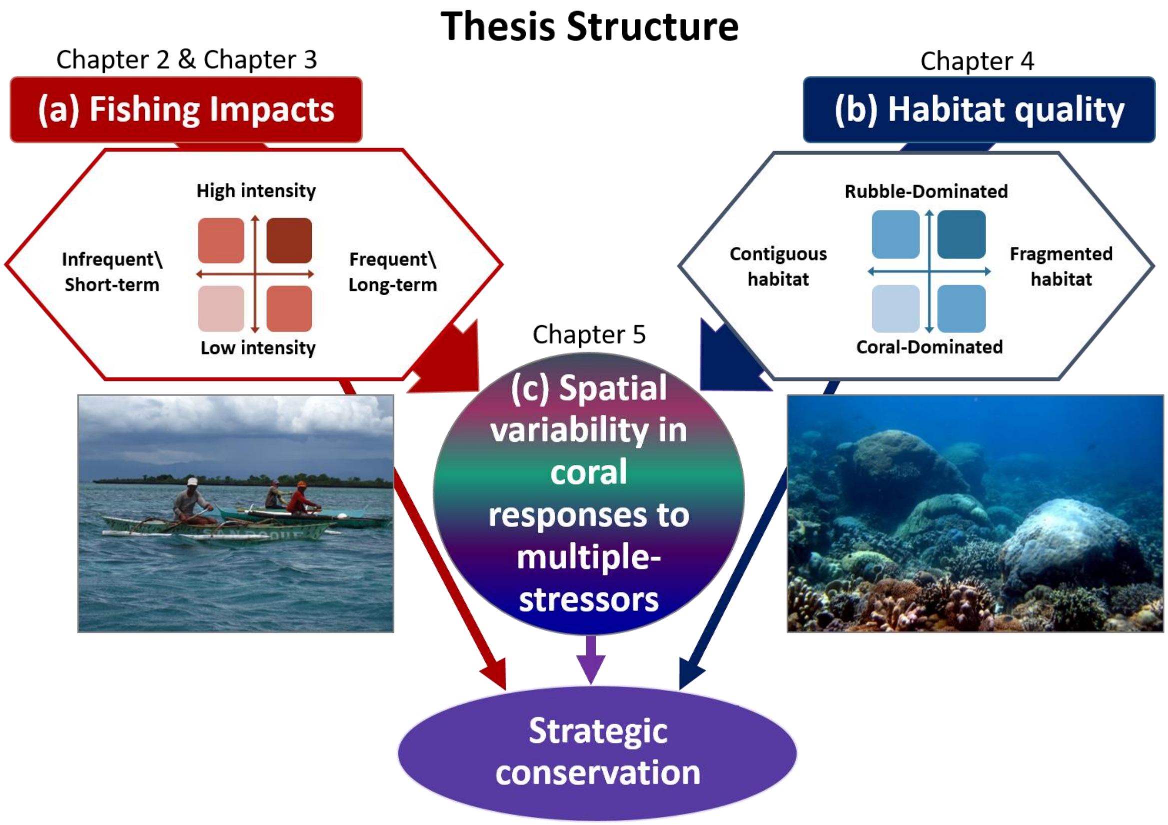

Figure 1.1 Schematic outline of dissertation.

I am particularly interested in this dynamic in small-scale fisheries where fishing communities are intimately connected to the resources they use. The composition of small-scale fisheries varies by region, but these fisheries use smaller boats and more basic technology relative to the industrial fisheries occurring in the same region. Among regions, however, small-scale fisheries have many things in common including: concentrated in inshore areas; cultural or ceremonial significance; limited management; low economic activity; minimally mechanized; and/or personal consumption or local sale of catches (Teh and Sumaila 2013). My study focuses on fisheries in a coral reef ecosystem in the Philippines. These reefs rank as a global conservation priority due to their combination of great biodiversity and acute threats.

While I was living in the Philippines, I became alerted to the work of the late Elinor Ostrom when she won a Nobel Prize. Her framework describing social-ecological systems (Ostrom 1990, 2009) was incredibly helpful in breaking down the complex problems I saw before me. Thus I expanded my original questions to evaluate how the sustainability of fishing practices was affected by governance. I explore these social-ecological interactions broadly at a system-scale, rather than in great detail at a site-scale. Addressing questions of ecological change at system-scales has become increasingly tractable due to the growing availability of high resolution satellite imagery. Scaling up what we know is becoming an increasingly important requirement for making research questions relevant to tackling global change.

1.2 Background and context

In response to rising pressures, the past few decades have seen a large number of efforts to protect the ocean. These wide-ranging initiatives have taken diverse approaches and occurred at all institutional levels, from local to global. Conservation efforts have been dominated by protected areas, which range in scope from the hundreds of small community-based marine protected areas (MPAs) in the Philippines to the approximately 1.5 million km^2 MPAs that currently are being established in both Hawaii and Antarctica. Beyond protected areas, there have been focused commitments to increase the sustainability of fisheries, improve ocean governance, create regional partnerships, reduce global carbon emissions, and regulate trade in threatened marine life (e.g. Convention on International Trade in Endangered Species (1975, with new marine species added starting in 2002); Coral Triangle Initiative (2009); FAO Voluntary Guidelines for Securing the Sustainability of Small-Scale Fisheries (2015); Sustainable Development Goals (2015); Paris Agreement on Climate Change (2016)).

Conservation initiatives have shown mixed results. In some cases where ecologically sound regulations were enacted there have been improvements in oceans. Fish stocks in the United States, for instance, have largely recovered from overfishing due to improved regulations (Hilborn 2007, Worm et al. 2009). Furthermore, there are growing examples where strong leadership, social cohesion, quotas, and appropriate incentives can facilitate successful governance of industrial and small-scale fisheries (Ostrom 2007, Gutiérrez et al. 2011). However, in other cases conservation initiatives have been less successful. Protected areas remain vulnerable to regional or global stressors (McMenamin et al. 2008, Graham et al. 2015). There can be extensive opposition to ocean protection (e.g. limits in catches or coastal development) from those who benefit from current unsustainable practices. Moreover, existing initiatives can be ignored by resource users or others for reasons ranging from economic gain, to poverty, to a lack of commitment or interest in outcomes (Ostrom 2009).

Of all stressors, fishing places the greatest pressure on the ocean, influencing marine life populations (fish, invertebrates), habitats, and ecological processes (Pauly et al. 1998, Worm et al. 2006, Burke et al. 2011). In 2010, for example, fisheries removed an estimated 94 million tonnes of marine life from the oceans (Pauly and Zeller 2016). Although these catches were dominated by industrial fisheries, catches from the small-scale sector – comprising 30% of the total – were not insignificant. Over the past half-century, industrial fisheries effort increased by 54% (Anticamara et al. 2011). During the same period, industrial fisheries expanded globally into the majority of highly productive areas of the ocean, including the high seas (Swartz et al. 2010). These changes have both direct and indirect influences on marine ecosystems. Fishing has extirpated populations of fish, particularly those that are high in the food chain, indiscriminately killed 10-27 million tons per year of non-target (bycatch) species, and extensively damaged marine habitats (Pauly et al. 1998, Watling and Norse 1998, Pauly and Zeller 2016). Consequences of these changes include reduced biodiversity, damaged food webs, and diminished ecosystem services (Pauly et al. 1998, Worm et al. 2006, Nañola et al. 2011).

Small-scale fisheries provide an excellent opportunity to examine the influence that fishing exerts on ocean ecosystems. There is much to learn about how small-scale fisheries interact with the environment, and much to be gained if these fisheries can be developed along sustainable pathways (FAO 2015). First, small-scale fisheries are incredibly important because they employ the majority of the world’s fishers and catch about half of the marine life (fish and invertebrates) that directly feed humans (Teh and Sumaila 2013, Pauly and Zeller 2016). Second, overfishing poses a great risk to the production and sustainability of small-scale fisheries (e.g. Stobutzki et al. 2006, Pomeroy 2012, Zeller et al. 2015). Third, small-scale fisheries tend to be poorly documented and challenging to govern (Johannes 1998, Jentoft and Chuenpagdee 2009). Small-scale fishing takes place at all hours of the day and are frequently spread out along coasts, rather than based in a central in a port. Developing species- or gear-specific management recommendations can be futile because small-scale fisheries – particularly in biodiverse coral reef ecosystems – catch a large variety of marine life and to do so use many fishing gears. Furthermore, many small-scale fisheries are found in developing countries, where natural resource management is restricted by limited funds and/or technical expertise.

Examining the evolution of small-scale fishing allows us to understand feedbacks between governance and sustainability of fishing practices. For coupled human-natural systems, governance can directly influence the choices of actors in the system, the status of the ecosystem, and the viability of the species that inhabit it. In fisheries, governance can draw upon a number of management tools and approaches including MPAs, constraints on who can participate, and restrictions on how participants can catch marine life. Such governance institutions and management tools change over time, reflecting dynamic societal priorities and values (Chuenpagdee and Jentoft 2015). However, the extent to which governance priorities influence how people will fish in the future is constrained by how they have fished in the past (i.e. the "memory" of the system) (Walker and Salt 2006). One broad trend in governance has been a shift in priorities. In the 1970s and 1980s, fisheries governance emphasized production (e.g. Asia Development Bank, World Bank, and Japanese government funding to Indonesia for tuna fishing) (Bailey and Jentoft 1990). More recently, governance has focused – in part – on sustainability, local empowerment, and food security (e.g. FAO Voluntary Guidelines for Securing Sustainable-Scale Fisheries).

In addition to increasing the effectiveness of fisheries governance, we need to identify and address other stressors on marine systems. The influence of human stressors on ecosystem change extends far back in time, varying from fire management to fishing (Jackson et al. 2001, Keeley 2002). These human imprints range from brash (e.g. exploitation of tar sands) to subtle (e.g. alterations to soil composition) (McKey et al. 2010, Rooney et al. 2012). The causal relationships of stressors are far more difficult to identify when they catalyze indirect changes or those that exhibit time lags (Walker and Salt 2006). Although ecosystems may recover once pressures are removed (Kittinger et al. 2011), the influence of humans can persist for centuries (Sutherland et al. 2016). For instance, the rapid increase in human pressures over the past century has sharply accelerated the rate of change into novel ecosystems (Lotze et al. 2006). In many cases, this forced change has greatly altered ecosystem structure, with the unintended consequence of reducing functioning and services. From these changes, we've learned that preventing damage is less expensive and more effective than trying to repair it (Sumaila 2004, Haisfield et al. 2010, Bekessy et al. 2010). Furthermore, quantifying the anthropogenic pressures that changed ecosystems can assist with identifying what pressures are sustainable, and also be used to identify targets for where to scale back (Jackson and Hobbs 2009).

To address a wide array of stressors, we need to know how they are distributed in space and time. Anthropogenic influences vary in frequency and intensity across ecosystems (e.g. with distance from communities). Some stressors are relatively stationary through time (e.g. point source pollution), while others change locations over time (e.g. popular fishing spots). Where multiple stressors overlap, their cumulative effects can be additive, antagonistic (less than their sum), or synergistic (greater than their sum). Thus, identifying where various stressors co-occur can be informative about ecological conditions. For small-scale fishing, targeted species of marine life are impacted by the spatial and temporal distribution of fishing effort and fishing gears. However, these fisheries can cause substantial collateral damage to habitats and ecosystems in heavily fished areas (Stobutzki et al. 2006, Mangi and Roberts 2006).

The effects of stressors vary across space due to underlying mosaics of biophysical conditions. Across marine ecosystems, conditions exhibiting spatial gradients include natural and anthropogenically influenced biophysical characteristics (e.g. current strength and direction, depth, habitat (type and configuration), pollution, temperature, water clarity). These features influence several aspects of marine ecosystems including larval settlement rates, mortality rates, and the physiological conditions of organisms (Almany 2004, Berkström et al. 2012, Olds et al. 2012). Spatial patterns of habitat configuration and abundance are particularly informative of a location's resilience to threats (Nyström and Folke 2001, Olds et al. 2012). Such landscape configurations influence the spatial distribution of energy, materials and species, the interactions between these elements, and temporal changes in these distributions and interactions (Turner 1989). In locations where human remove, degrade, or fragment ecologically valuable habitats (Brooks et al. 2002), habitat loss and fragmentation can increase predation risk and lower genetic flow among sub-populations of threatened organisms (Andren and Angelstam 1988, Saunders et al. 1991, Selgrath et al. 2007).

Knowledge about a system can be gathered from several sources including scientific inquiry and local environmental knowledge (LEK). Science acquires knowledge through experimental manipulations or surveys measuring variations in conditions. The spatial scale of surveys range from small study areas (< 1 km^2), to ecosystems, to the entire planet, with decreasing resolution at broader scales. Scientific conclusions are derived from statistical analyses of information which evaluates the probability and strength of various relationships (McNie 2007). Conservation planning can also be supported by LEK (Hind 2015). This type of knowledge comprises the integrated and situated knowledge, beliefs, and practices of communities and resource users regarding the local environment and their relationship with it (Berkes 2012, McMillen et al. 2014). Often gathered through interviews, LEK has great potential for improving conservation and management by increasing knowledge, complementing scientific measurements, and informing conservation strategies (Thornton and Scheer 2012). One form of LEK, participatory mapping, can be used to evaluate the spatio-temporal distribution and dynamics of social-ecological features, such as fishing pressure.

It is rare to integrate thinking about fishing pressure, other stressors, and associated ecological conditions across both time and space. There are excellent examples of these components being analyzed individually or in a limited combination (Worm et al. 2006, Darling et al. 2010, Swartz et al. 2010, Ban et al. 2014, Halpern et al. 2015, Cinner et al. 2016, Côté et al. 2016, Pauly and Zeller 2016). For instance, there is a growing awareness that fisheries management can be improved using spatial information (e.g. improving the accuracy of catch estimates; quantifying the collateral damage caused by fishing) (Walters 2003, Stewart et al. 2010, Swartz et al. 2010). At the same time, there have been impressive global efforts to reconstruct the recent history of global fisheries, cumulating in the recent estimation of global catches from 1950-2010 (Pauly and Zeller 2016). These reconstructions are deeply informative of how pressures on the ocean have changed over time. They do not, however, consider the spatial distribution of fishing impacts, fishing interaction with other stressors, or ecological effects. Moreover, assessments of multiple stressors frequently infer fishing impacts rather than quantifying effects – in part because fisheries are difficult to manipulate experimentally (Ban et al. 2014). For instance, spatial estimates of changing impacts to oceans have not yet attempted to evaluate the effects caused by cumulative impacts, but have instead focused on the changing intensities of stressors (Halpern et al. 2015). My thesis aims to fill this gap by evaluating how the state of corals corresponds to the spatial distribution of long-term fishing pressure, other stressors, and biophysical conditions.

1.3 Case study

In this thesis, I examine how ecosystems respond to small-scale fishing and other pressures over space and time, using a mixed-methods case study in the central Philippines. The case study method focuses on the empirical inquiry of individual situations to explore 'how' and 'why' questions about contemporary phenomenon in social systems (Yin 2013). I selected the Danajon Bank in the central Philippines (Figure 1.2) as a case study for several reasons, including: conservation and poverty concerns, the region's long-term reliance on fishing, the exclusion of commercial fisheries (enabling the examination of small-scale fisheries without the influence of commercial fisheries), and the extremely large number of fishers in the region (increasing the number of actors and mechanisms) (Flyvbjerg 2011).

The Philippines is consistently ranked as a global conservation priority country because of its combination of high biodiversity and high threats (Roberts et al. 2002, Selig et al. 2014). The country supports about 22,000 km^2 of coral reefs and contains a global center of marine biodiversity, with high levels of endemic and widespread species throughout (Allen 2008).

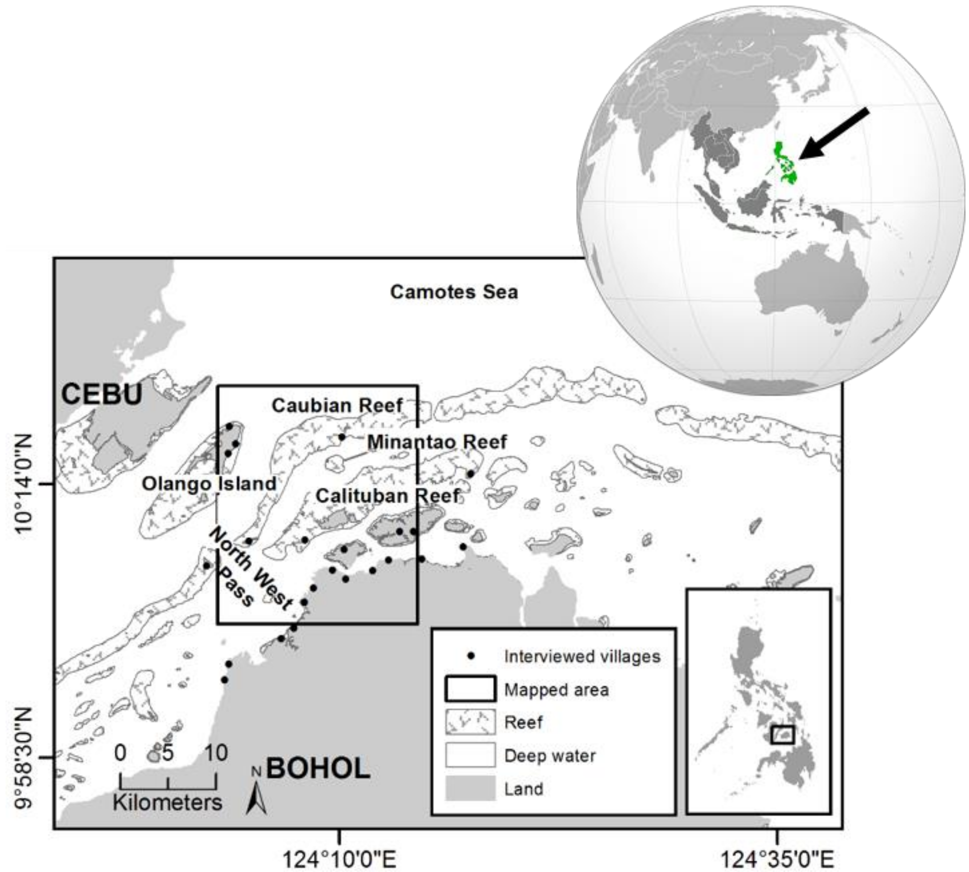

Figure 1.2 Danajon Bank study area was located in the central Philippines.

These high levels of biodiversity have emerged from a combination of geologic, geographic, and oceanographic circumstances, including high levels of heterogeneity (Sanciangco et al. 2013).

As a nation, the Philippines is highly dependent on the ocean. According to the FAO, the Philippines ranks as the 12th largest producer of marine capture fisheries despite having lower technical capacity than several leading countries (e.g. China, United States, Russian Federation) (FAO 2016). However, the Philippines may in fact rank much higher because the FAO routinely underestimates catches from small-scale fisheries – a major component of Philippines fisheries (Pauly and Zeller 2016). Philippine marine biodiversity is also threatened by human activities on land including erosion and runoff from poorly planned development, extensive logging, and mining. Additionally, their seas face high levels of nutrient pollution from untreated sewage and agriculture (Burke et al. 2011).

The Philippines has taken notable ocean-focused conservation actions, among them the regulation of fishing sectors and fishing gears. Although it was not solely focused on conservation, national legislation (1998 Fisheries Code: RA 8550) contains conservation-based components such as banning all fishing gears causing habitat destruction and many active fishing gears. In some cases, mesh sizes were also limited. Additionally, there has been a large focus on marine protected areas (MPAs) as a conservation tool (Martin-Smith et al. 2004, Weeks et al. 2010, Horigue et al. 2012).

In the Philippines, small-scale fisheries are referred to as 'municipal fisheries' and are restricted to using boats weighing less than 15,000 gross tons. In the Philippines, these fisheries are renowned for their diversity of fishing gears (Umali 1950), but it is not known how the use of fishing gears has varied over time. Additionally, changes in the effort and extent of fishing pressure are not well documented.

I worked in the Danajon Bank (10˚15'0'N, 124˚8'0'E), a double barrier reef in the Central Visayas, Philippines. This ecosystem sits off the northern and northwestern coasts of Bohol – a rural province that exemplifies many of the challenges facing the country as a whole. The region struggles with extreme poverty and minimal infrastructure. Between 45% and 70% of the residents live below the Philippines poverty level (Provincial Government of Bohol 2011) and island villages generally lack electricity, running water, and septic systems. Fishing is a primary human activity influencing the Danajon Bank because of its large marine area (235,242 ha), high population density, and lack of alternative livelihoods. Although its communities are predominantly rural, the Danajon Bank lies very close to Cebu City, the second largest metropolitan area in the Philippines (with 2.8 million people). Since the 1998 Fisheries Code, communities in the Danajon Bank have established over 40 locally-enforced no-take marine protected areas (MPAs). Although municipalities are official responsible for establishing MPAs, that process has largely been driven by fishing communities (Ban et al. 2009). Despite these management efforts, heavy fishing pressure, destructive fishing, and high population densities continue to apply huge pressure. Conservation efforts thus far may have managed to slow the rate of decline, but they certainly haven't led to recovery (Christie et al. 2006, Marcus et al. 2007).

My thesis focused on a 418 km^2 (19 km by 22 km) area in the central Danajon Bank. I conducted interviews in 23 fishing communities and randomly sampled fishers from within those communities. For fishing, I considered historical changes as far back as my respondents could remember, giving me the capacity to evaluate changes from 1950/1960 to 2010. This half-century was characterized by rapid human population growth in the region (Philippines National Statistics Office 2010), and declines in both catch-per-unit-effort (CPUE) and biodiversity (White and Cruz-Trinidad 1998, Nañola et al. 2011). As such, understanding these spatial long-term trends has both social and ecological implications. Additionally, I conducted or supervised biological surveys supporting habitat mapping across the Danajon Bank. One of the goals of my broad and historical project is to pull together information to assist the region's active conservation planning efforts.

1.4 Context for the field study

In the Philippines, my research was supported by ongoing collaborations with the Project Seahorse Foundation for Marine Conservation (previously part of Haribon Foundation for Conservation of Natural Resources and now ZSL Philippines). This Filipino NGO has led work in community-based conservation in the Danajon Bank ecosystem, where my study took place, for more than two decades. The group has made notable contributions in scientific research, fisher empowerment, and regional governance. As one key example, they have led on establishing over 30 community-based MPAs, with a great deal of community organizing and technical input. During my fieldwork, PSF helped me establish contacts with government leaders (mayors, village captains) across the Danajon Bank and provided me with guidance and resources for working in fishing communities.

1.5 Research questions

In this thesis, I focus on four main research questions addressing conservation for data-poor fisheries and coral reefs. These questions are as follows:

1) How has the use of fishing gear changed, and how were these changes correlated with fisheries governance? (Chapter 2)
2) How have the spatial dimensions of fishing effort changed? (Chapter 3)
3) How can we map coral reef habitats quickly and effectively? (Chapter 4)

4) Can we use the spatial distribution of anthropogenic stressors and biophysical conditions to predict the presence of living corals? (Chapter 5)

1.6 Thesis outline

This dissertation contains this introduction (Chapter 1), four chapters based on novel data (Chapters 2-5), followed by a general discussion about lessons learned, further questions, and applications for conservation biology (Chapter 6).

1.6.1 Chapter 2: Shifting gears: Sixty years of diversification, intensification, and effort increases in small-scale fisheries

In this chapter, I evaluated how small-scale fishing has changed over time, and the influence of evolving fisheries governance on such transitions. For fisheries, over-exploitation creates problems such as reduced biodiversity and lower catches. I focused on fishing gears because gears influence fisheries' impacts on ocean ecosystems and gears are often the focus of governance measures. For this chapter, I used fisher interviews to characterize 60 years (1950–2010) of fishing gear dynamics. I asked two questions. (1) Are historical trends in fishing gear use indicative of intensification? (2) Do fishing gear trends parallel changes in fisheries governance priorities? I also evaluated changes in fishing intensity as reflected by the use of destructive, active, non-selective, and illegal gears.

1.6.2 Chapter 3: Incorporating spatial dynamics greatly increases estimates of long-term fishing effort: A participatory mapping approach

In Chapter 3, I investigate the spatial and temporal dynamics of small-scale fisheries in the Danajon Bank and evaluate how estimates of fishing pressure are influenced by non-spatial vs. spatial measures of change. The spatial distribution and intensity of fishing is dynamic through time and shapes many ecosystems. In coral reefs, fisheries frequently are purported to cause declines in both biodiversity and catches. However, long-term spatial data quantifying how fishing effort and fishing gears might provoke such declines are often missing. For this chapter, I used participatory mapping to evaluate the spatio-temporal dynamics of small-scale fishing in a 418 km^2 area of the Danajon Bank over half a century (1960–2010). I address three questions. (1) How do estimates of the change in fishing effort over time vary between non-spatial and spatial metrics? (2) How have the spatial extent, frequency, and intensity of fishing effort changed in a locally fished area? (3) How can maps of historic fishing pressures be used to strengthen conservation and management?

1.6.3 Chapter 4: Mapping for coral reef conservation: Comparing the value of participatory and remote sensing approaches

In Chapter 4, I developed habitat maps of coral reef ecosystems while exploring the conservation tradeoffs between using participatory mapping and remote sensing. Detailed habitat maps are critical for conservation planning, yet for many coastal habitats only coarse-resolution maps are available. As the logistic and technological constraints of habitat mapping become increasingly tractable, habitat map comparisons are warranted. Here I compared two mapping approaches: local environmental knowledge (LEK) obtained from interviews; and remote sensing analysis

(RS) of high spatial resolution satellite imagery (2.0 m pixel) using object-based image analysis. In this chapter I focus on three questions. (1) How accurate are the two mapping techniques for mapping shallow seafloor habitats? (2) How does each mapping approach depict the habitat area and seascape characteristics of the ecosystem? (3) What are the conservation implications and costs that influence which mapping approach is most appropriate for different situations?

1.6.4 Chapter 5: Synergistic stressors offset by depth, management, and landscape structure in a coral reef ecosystem

In chapter 5, I characterize locations within a coral reef ecosystem that have withstood or adapted to local stressors as indicated by the dominance of living coral. The influence of multiple stressors on ecosystems has proven difficult to quantify and presents a key challenge for conservation. For this project I integrated the high spatial resolution LEK maps of historical fishing (Chapter 3) and the satellite-imagery maps of benthic habitats (Chapter 5). I then identified anthropogenic and biophysical characteristics of the system that were correlated with the presence of living coral. Corals are particularly good for exploring long-term ecosystem status because when corals are stressed and die, physical evidence of their prior distribution can still be documented. I asked three questions. (1) How is the spatial distribution of living coral related to anthropogenic stressors and biophysical conditions? (2) Do multiple stressors affecting corals have additive, synergistic, or antagonistic impacts? (3) Over what time-scale do corals respond to fishing pressure (1960-2010)?

1.6.5 Chapter 6: Discussion

In Chapter 6, I end with a general discussion of findings from my research and consider both conservation implications and future research directions.

To consider the questions outlined above, I integrated diverse approaches. It remains perplexing to understand interactions between the human and environmental components of stressed systems. The large number of variables and interactions involved are challenging to follow and frequently require unrelated tools and skills to unravel. Techniques for addressing individual components of these systems, however, do exist. Thus, there is great opportunity for supporting conservation through transdisciplinary research. This practice weaves together the methods of different disciplines to develop a more holistic knowledge of the characteristics and challenges facing a system. In taking this approach, I aimed to incorporate new technologies (e.g. high spatial resolution satellite imagery) and to push the boundaries of what information established methods could provide (e.g. covering a long time frame using participatory mapping). I hope that my research will contribute to solutions to the long-enduring problems considered in my thesis.

Chapter 2: Shifting gears: Sixty years of diversification, intensification, and effort increases in small-scale fisheries

2.1 Summary

Locally sustainable resource extraction activities, at times, transform into ecologically detrimental enterprises. Understanding such transitions is a primary challenge for conservation and management of many ecosystems. In marine systems, over-exploitation of small-scale fisheries creates problems such as reduced biodiversity and lower catches. However, long-term documentation of how governance and associated changes in fishing gears may have contributed to such declines is often lacking. Using fisher interviews, I characterized fishing gear dynamics over 60 years (1950–2010) in a coral reef ecosystem in the Philippines subject to changing fishing regulations. Most individual fishers used one or two gears at a time (mean number gears < 2 in all years). In aggregate, however, fishers greatly diversified their use of fishing gear types. Individual fishing effort was fairly steady over the study period, but total fishing effort by all fishers increased 240%. In particular, I document large increases in the total effort by fishers using nets and diving. Other fishing gears experienced less pronounced changes in total effort over time. Fishing intensified through escalating use of non-selective, active, and destructive fishing gears. I also found that policies promoting higher production over sustainability impacted the use of fishing gears, with impacts persisting decades after those same policies were stopped. My quantitative evidence shows dynamic changes in fishing gear use over time and indicates that gears used in contemporary small-scale fisheries impact oceans more than those used in earlier decades.

2.2 Introduction

Long-term assessment of natural resource exploitation is critical to understanding ecosystem change and how governance affects those changes (Gelcich et al. 2010). Contemporary ecosystems are shaped by their history of disturbances and human interventions (Christensen 1989). These historical impacts influence the distribution of species and habitats that we see today and can affect modern-day ecosystem functioning and services (McKey et al. 2010, Tomscha and Gergel 2016). In addition, the historical context can inform managers and other actors tasked to understand current conditions and to direct future ecological change (Thurstan and Roberts 2010, McClenachan et al. 2012).

The need for a grounded historical perspective and the integration of non-traditional data sources are particularly apparent for small-scale fisheries (Jackson and Hobbs 2009, McClenachan et al. 2012, Hind 2015). Although small-scale fisheries use small boats with basic fishing equipment, they represent a critical component of global fisheries. Small-scale fisheries collectively employ over 230 million people in the direct and indirect sectors (Teh and Sumaila 2013) and have edible catches rivaling those of industrial fisheries (Pauly and Zeller 2016). Widespread overfishing has led to international commitments to improve their sustainability (FAO Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries) (FAO 2015) and a call for research to assess the effectiveness of management approaches (Johnson et al. 2013). Although catch declines in these fisheries are often attributed to the intensification of fishing activities, little long-term information exists to contextualize or quantify these changes (Jennings and Polunin 1996). In many cases, the trajectories of small-scale fisheries remain undocumented because they involve a large number of fishers (25 times more individuals than industrial fisheries), are decentralized, and occur in countries with limited governance, funds, and/or technical expertise (FAO 2015). In such data-poor systems, local environmental knowledge (LEK) can provide information about the history of ecosystems and the practices of humans who depend on them (Hind 2015).

Evaluating the choice of fishing methods and associated gears – such as fish traps and drift-nets – deployed over time is one key to understanding small-scale fisheries (Marchal et al. 2007). Small-scale fisheries use many gears and catch a plethora of fish and invertebrate species (e.g. cephalopods, gastropods). Fishers adapt gear usage for changing biophysical conditions (e.g. tides, depth), different habitats (e.g. coral, seagrass), as well as to incorporate species behavior (Ruddle 1996). Additionally, fishers can change gears and adjust the effort allocated to various gears in response to the availability of marine life, evolving biophysical conditions, and market competition (Ruddle 1996, Marchal et al. 2007, Anderson et al. 2011). Fishing gears can affect both the diversity and the abundance of species in different ways (Lokrantz et al. 2010). Some gears damage marine habitats (Mangi and Roberts 2006) and others catch non-target and juvenile marine life (i.e. bycatch) (Hicks and McClanahan 2012). Thus, trends in gear use can be used to infer trends in fishing impacts to ecosystems.

Approaches to managing and governing small-scale fisheries are vast, but often include gear restrictions and protected areas (Ban et al. 2009a). Management as well as institutional approaches to governance change over time, reflecting dynamic societal priorities and values (Chuenpagdee and Jentoft 2015). For example, governance that prioritizes the goals of a central government frequently maintains hierarchical management institutions. In contrast, governance that values local empowerment may develop co-management institutions based on local participation (Wamukota et al. 2012). In many countries, institutional priorities have swung from an emphasis on extraction and resource exploitation in the 1970s and 1980s (Bailey et al. 1986) to a contemporary focus on sustainability (BFAR 2015, FAO 2015). The extent to which resource users change their practices in step with governance priorities is influenced by how strongly past practices constrain and guide future choices (i.e. the “memory” of the system) (Walker and Salt 2006).

The need for effective management is particularly relevant in small-scale fisheries targeting coral reefs in the Philippines, a country of global priority for biodiversity conservation (Selig et al. 2014). Coral reefs are among the most productive and biologically diverse ecosystems in the world, but face many stressors including climate change, destructive fishing, overfishing, and pollution (Moberg and Folke 1999, Hughes et al. 2003). At the apex of the Coral Triangle, the Philippines is located in the global epicenter of marine biodiversity (Carpenter and Springer 2005) and contains the third most extensive reef system in the world (about 22,000 km^2) (Wilkinson 2000). The country is highly dependent on fishing for food and livelihoods (FAO 2016). Reef fisheries in the Philippines once provided up to 37 tons per km^2 per year, but catches throughout the country have sharply declined (Gomez et al. 1994, Green et al. 2004, Stobutzki et al. 2006). Due to its combination of high biodiversity and high threats, the management of Philippine small-scale fisheries is an important challenge globally.

In this study, I characterized 60 years of fishing gear dynamics in a small-scale fishery in the central Philippines. I examine a time period (1950–2010) coinciding with extensive changes in fisheries governance, ecosystem degradation, declining catches, and increasing populace (Batongbacal 2002, Marcus et al. 2007, National Statistical Coordination Board 2010, Nañola et al. 2011). I ask two questions. First, are historical trends in fishing gear use indicative of intensification? Second, do fishing gear trends parallel changes in fisheries governance priorities? As no long-term records of fishing practices exist in our study area, I used local environmental knowledge to document the history of fishing gear use, including changes in the diversity of gears in use, as well as fishing effort. I also evaluated changes in fishing intensity as reflected by the use of destructive, active, non-selective, and illegal gears.

2.3 Methods

2.3.1 Study site

I focused on small-scale fisheries in the Danajon Bank (Central Visayas, Philippines; 10˚15’0’N, 124˚8’0’E; Figure 2.1). This coral reef ecosystem is characterized by large proportion of degraded habitats (Selgrath et al. 2016) and exceptionally low biomass of fish (e.g. demersal fish biomass 0.45 tons/sq km) (Fisheries Improved for Sustainable Harvest (FISH) Project 2010)). The Danajon Bank sits off the northern and northwestern coasts of Bohol – a province that struggles with extreme poverty and minimal infrastructure (PPDO Bohol 2013), traits that are typical of places supporting small-scale fisheries. Fishing is a primary human activity influencing the Danajon Bank because of the high population density and lack of alternative livelihoods. The Danajon Bank lies very close to Cebu City, the second largest metropolitan area in the Philippines (with 2.8 million people). However, the Danajon Bank’s communities are rural and travel links to the city are weak.

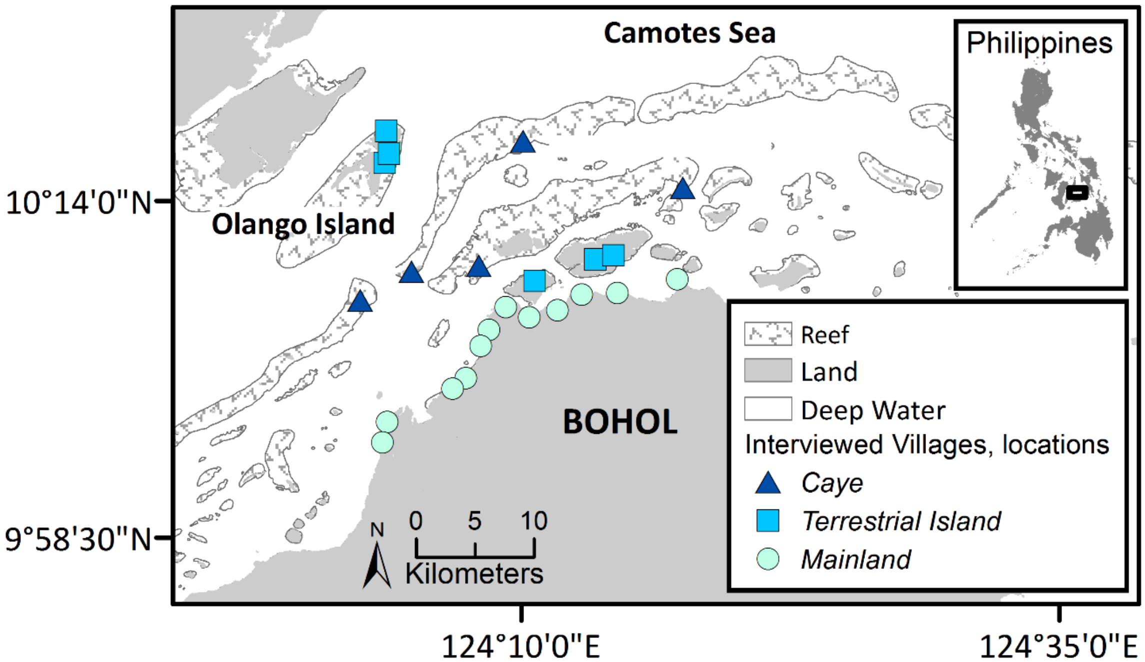

Figure 2.1 Map of the study area and 23 sampled villages in the Danajon Bank, Philippines, a biodiversity hotspot within the Coral Triangle. Symbols indicate the location of villages in the ecosystem.

Contemporary fisheries management in the Philippines emphasizes co-management, gear restrictions, spatial restrictions, and marine protected areas (MPAs) (Batongbacal 2002, Weeks et al. 2010). In the Philippines, small-scale fishers are legally known as ‘municipal’ fishers. To participate in these fisheries, boats must weigh 3 gross tons or less. Most fishers use outrigger canoe style boats, but engine-use has increased from approximately 20% of boats in 1960 to 50% of boats in 2010 (Selgrath et al. 2017a). Small-scale fisheries in the Danajon Bank are multi-gear, multi-species, and effectively open-access (Christie et al. 2006). Filipino laws have given small-scale fishers exclusive rights to fish in inshore waters < 15 km from coasts since 1991. The entire Danajon Bank falls within inshore waters. Despite ongoing management efforts, heavy fishing pressure, destructive fishing, and high population densities continue to apply huge pressures on marine ecosystems (Christie et al. 2006, Marcus et al. 2007). Our study focused on an 800 km^2 section of the central Danajon Bank. This area spanned five municipalities, two provinces, and a gradient from inshore turbid waters to offshore clear waters.

2.3.2 Eras of fisheries governance

Six decades were examined in our study and partitioned into governance eras based on extensive review of government fisheries documents, legislation and development projects, and other published and online sources. I focused on the three aspects: (i) level of organization and power, (ii) aims and values of fisheries legislation, and (iii) aims and values of development funding. Each was evaluated according to its management tools, institutional formation, as well as underlying principles and values, thereby providing information on different aspects of governance (Chuenpagdee and Jentoft 2015). Recent participation in co-management was characterized by: (i) presence of community-established marine protected areas (MPAs) and (ii) participation in fisher organizations (details in Fisher Interviews).

2.3.3 Fisher interviews

To document temporal changes in fishing activities, I drew on LEK by conducting semi-structured interviews between July 2010 and April 2011 (n=391 fishers) in 23 fishing communities (approximately 50% of the communities in the study area; Table 2.1). The participation rate of fishers was approximately 100%. See Appendix A for interview questions. LEK integrates the customary knowledge, practices, and beliefs of communities regarding the local environment and their relationship with it (Berkes 2012, McMillen et al. 2014). I stratified

Table 2.1 Summary information about 23 fishing communities in the central Danajon Bank, Philippines including population sizes, sample sizes, and the presence of fishers' organizations and MPAs. Ter = terrestrial.

Province	Municipality	Location	Village	Village Population (2010)	No. Fishers in Village (2010)	No. Fishers Interviewed (2010)	Fisher Organization (Presence)	MPA (Year Started)
Bohol	Buena Vista	Caye	Cabul-an West	1884	349	28	1	2000
Bohol	Buena Vista	Mainland	Asinan	757	71	5	1	2000
Bohol	Buena Vista	Mainland	Cruz	745	38	9	1	none
Bohol	Getafe	Caye	Nasingin	1441	335	25	1	2002
Bohol	Getafe	Caye	Pandanon	1833	320	21	1	2002
Bohol	Getafe	Ter. Island	Jandayan Norte[a]	694	23	11	1	2002
Bohol	Getafe	Ter. Island	Mahanay	448	21	2	1	2003
Bohol	Getafe	Mainland	Carlos P. Garcia	725	21	1	0	none
Bohol	Getafe	Mainland	Corte Baud	622	82	4	1	2002
Bohol	Getafe	Mainland	Poblacion	2210	36	3	1	none
Bohol	Getafe	Mainland	San Jose	1441	50	3	0	none
Bohol	Getafe	Mainland	Tugas	817	58	4	1	2002
Bohol	Getafe	Mainland	Tulang	1594	75	6	1	2003
Bohol	Inabanga	Mainland	Lawis	2629	447	31	1	2000
Bohol	Inabanga	Mainland	Ondol	1098	131	9	1	2000
Bohol	Talibon	Caye	Calituban	4527	552	37	1	2000
Bohol	Talibon	Ter. Island	Mahanay	2012	113	8	1	unclear
Bohol	Talibon	Mainland	Bagacay	3106	213	16	0	none
Bohol	Talibon	Mainland	San Francisco	5870	408	25	1	1996
Cebu	Lapu Lapu	Caye	Caubian	2114	429	30	1	2007
Cebu	Lapu Lapu	Ter. Island	Baring	2934	530	37	0	2003
Cebu	Lapu Lapu	Ter. Island	Talima	4945	829	58	0	2003
Cebu	Lapu Lapu	Ter. Island	Tungasan	1754	242	18	0	none

fishing communities by their location (Coastal, Terrestrial Islands, Cayes) (Ban et al. 2009b); Figure 2.1) to ensure I captured representative samples of fishing gears adapted to local environmental differences (e.g. mangroves vs. reefs). I then proportionally sampled randomly-selected communities from within each group using a randomly-generated numbered list. I obtained written consent from municipal mayors, in accordance with Philippine laws, and oral consent from elected village officials, as well as every individual respondent. When oral consent was given by respondents, it was recorded on consent forms. Oral consent was most appropriate culturally and preferable due to the low levels of education and literacy among some participants. Research methods, including consent procedures, were approved by the University of British Columbia's Human Behavioural Research Ethics Board (H07-00577).

Within each participating community, I identified fishers using village-level census data. Throughout the Philippines, census records are collected regularly by community health workers. Where occupational data was incomplete, outdated, unclear, or had questionable accuracy (e.g. 25 year olds listed as high school students), health workers provided missing information. For example, health workers identified those fishers who had moved, passed away, or changed livelihoods. I used village census records rather than municipal records of fishers because I found that municipal data severely underestimated the number of all people who fished. Census occupational data also provide a conservative estimate of fishers because these records do not include children or women who fish (Kleiber et al. 2014). I did not attempt to correct for this bias.

After identifying fishers within participating communities, I randomly sampled 7% of fishers. To obtain longer time series of information, I stratified fishers by age, focusing on fishers born before 1981 whom I estimated had been fishing for at least 15 years. This method may underrepresent recent trends influenced by young fishers, but prioritizes a long time-series of fishing. Respondents provided oral consent to participate in this research, as is culturally most appropriate in these villages. Interviews were conducted in the local language (Cebuano) by two Cebuano-fluent local research assistants.

During interviews, I systematically reconstructed the history of the respondent's fishing practices, as far back as the respondent could remember, using three steps. First, I constructed personal timelines for each respondent which helped fishers link reported changes in fishing practices with important dates (e.g. age started fishing; date of children's births) (Means and Loftus 1991). Second, I determined which years each respondent fished in the Danajon Bank (either full- or part-time), as well as any monthly or migratory patterns in his or her fishing. Also, the days per year that a respondent fished in our study area was used to quantify individual fishing effort. As some respondents migrated to other provinces for several months a year, I excluded any fishing activities outside of the Danajon Bank from this study. Third, I made timelines for each fishing gear that each respondent used in our study area. For each gear, I recorded the years the respondent fished with the gear and any details he or she shared about the fishing gear (e.g. catch). During interviews I validated the consistency of fisher responses using internal triangulation. I asked questions about key information during multiple questions, which allowed us to cross-check the consistency of fishers' responses. This enabled us to correct inconsistent or illogical answers during interviews (Mathison 1988). I focused on fishing gears and effort rather than catches as these tend to be relatively consistent over time. This consistency makes gears and effort easier to recall than variable aspects of fishing, such as catches (Neis et al. 1999, O'Donnell et al. 2012)

2.3.4 Indicators of co-management participation

Fishers' organizations are one official component of Philippine local co-management structure, as outlined by the 1998 Fisheries Code (RA 8550). I sought to evaluate the presence of these organizations, and the level of fisher engagement. I asked a subset of respondents (n = 256 fishers) whether fishers' organizations existed in their communities, whether they participated in the fishers organizations, and the year that they had joined the organizations. I also used expert interviews with local community organizers to confirm information about when fisher organizations were established. Since MPAs are a second component of local fisheries co-management (RA 8550), I identified which communities had established MPAs through expert interviews, websites of municipal governments, and published literature (Alcala et al. 2008).

2.3.5 Fishing gear classifications

For gear classifications, I developed a database based on information derived from surveys as well as from key informants, our own field knowledge, and published literature on fishing gears (Umali 1950, Alcala and Gomez 1987, FAO 1990, Ruddle 1994, Jennings and Polunin 1996, Green et al. 2000, 2004, Dugan et al. 2003, Mangi and Roberts 2006, Guieb 2008, Shester and Micheli 2011). I assigned each specific gear to one of eight general gear classes that I set to fit the small-scale fisheries context (Table 2.2; Appendix B): hook & line, nets, diving, traps, blast fishing, poison, fish corrals, and gleaning. For example, hook-and-line methods include gears with various numbers of hooks, lines, and weights.

Table 2.2 The eight general classes of fishing gears and some examples of specific gears and their classification in intensive gear categories. Intensive categories indicate if the gears are: active, destructive, non-selective, and/or illegal.

General Gear Class	Specific Gear	Intensive Categories			
		Non-selective	**Active**	**Destructive[a]**	**Illegal[b]**
hook & line	hand line, 1 hook				
	benthic longline				
nets	bottom set gillnet	X			
	encircling gillnet	X	X		1998 (if automated when deployment from boats)
diving	spear	X	X		
	crowbar (Cebuano: *kay-kay)*	X	X	breaks corals	1998
traps	large fish traps				
	crab traps				
blast fishing	fertilizer bomb	X	X	shatters corals	1932
poison	squirt bag	X	X	kills corals	1932
	in traps	X		kills corals	1932
fish corral	v-shaped weir				
gleaning	hand	X	X		
	machete	X	X	breaks corals	1998

[a]Brief descriptions of habitat impacts are included for destructive gears.
[b]For illegal gears I provide the year that the gear became illegal and any conditions about the legality, if relevant.

In coral reefs, fishing gears span a range of intensities, from hand-reeled lines that catch one fish at a time (handlines), to homemade explosives that kill entire schools of fish and shatter adjacent corals (blast fishing). I define intensive fishing gears as those with a high magnitude of force, an avaricious efficiency, and/or a relatively low selectivity. In this way, intensive gears may have higher catchability than non-intensive gears, but they are also likely to cause greater impacts to the ecosystem (e.g. via damaging habitat, catching juveniles). Since the intensity and impacts from small-scale fishing gears have rarely been quantified, this approach provides a process for evaluating the potential impacts of gears (but see, for example, (Mangi and Roberts 2006, Shester and Micheli 2011, Hicks and McClanahan 2012) for small-scale fisheries).

I assigned the 93 specific fishing gears into four pairs of intensive/non-intensive categories: destructive/non-destructive; active/passive; non-selective/selective; and illegal/legal (Appendix B). Destructive gears damage habitats (e.g. blast fishing). Actively moving gears heavily exploit marine life (e.g. trawling) and raise concerns in fisheries conservation (for example, (Tillin et al. 2006)). Non-selective gears catch juveniles and non-target species (e.g. small-mesh nets) (Hicks and McClanahan 2012). Illegal gears may be intensive or may be prohibited for other reasons (e.g. dangerous for fishers), and are defined by regulations rather than by impact. Some gears were illegal throughout the study period (e.g. blast fishing), while other gears became illegal during the study period (e.g. fishing with a crowbar which damages habitat and thus became illegal under the 1998 Fisheries Code). I placed each specific gear in only one category within each pair (i.e. destructive *or* non-destructive), but a fishing gear could belong to more than one intensive category (e.g. beach seines are destructive, active, and non-selective).

2.3.6 Data analyses

2.3.6.1 Trends in individual and aggregate fishing gears

First, I calculated the mean number of specific gears that individual fishers had used throughout their careers. Second, I evaluated how the mean number of specific gears that individual fishers used in a year changed over time. Third, I assessed how two metrics of gear diversity had changed over time for specific gears: gear richness and the Simpson's Index of Diversity (Peet 1974). Gear richness (G; hereafter diversity) was estimated as the total number of unique gears (*g*) used in a year (*t*)

$$G = \sum_{1}^{f} g_t \qquad (2.1)$$

The Simpson's Index of Diversity (D) considers both gear richness and evenness by estimating the probability that two gears taken at random will represent the same gear. I estimated D where N_t is the total number of any type of gear used by all fishers in year (*t*) and n_t is the total number of gears of a particular type of gear used by all fishers in a year (*t*)

$$D = 1 - \frac{\sum_1^G n_t(n_t - 1)}{N_t(N_t - 1)} \qquad (2.2)$$

For analyzing these gear changes over time, I divided the fishing timelines into the four governance eras and sampled gear-use during six randomly selected years from each era. The residuals from the Bartlett test comparing changes in in gear use and diversity were non-normally distributed, so I used Kruskal-Wallis Rank Sum tests to compare differences in gear use between governance eras (Crawley 2012). I used a post-hoc Kruskal-Wallis Multiple Comparison test to determine which governance eras exhibited significant differences in gear use and diversity.

2.3.6.2 Trends in general, intensive and non-intensive fishing

To understand trends in fishing activities, I evaluated how three aspects of fishing developed over time and in relation to governance eras: (i) total fishing effort (fishing days per year); (ii) relative fishing effort (i.e. the proportion of total fishing effort allocated to each gear); and (iii) the proportion of fishers using various fishing gears. I assessed the evolution of these three aspects of fishing for both the eight general fishing gear categories and the four (non-exclusive) pairs of intensive and non-intensive gears. Estimates for total fishing effort (days per year) assess the fishing effort for only the 23 participating communities.

For each category, I estimated effort and gear use using six steps. First, I used interview data to calculate the total days per year (effort) that respondents fished as the sum of individual effort (*e*) by all respondents (*f*) in year (*t*)

$$\hat{E}_t = \sum_{1}^{f} e_{ft} \qquad (2.3)$$

Second, I evaluated the mean fishing effort in a year ($\bar{E}_t$) as total days fished ($\hat{E}_t$) in a year (*t*) by all respondents (*f*)

$$\bar{E}_t = \frac{\hat{E}_t}{f} \qquad (2.4)$$

Third, for each year I divided individual respondents' fishing effort among their actively used fishing gears. Fourth, I estimated gear-specific effort as the sum of effort (*e*) by individual respondents (*f*) using each gear (*g*) in each year (*t*)

$$\hat{E}_{gt} = \sum_{1}^{f} e_{fgt} \qquad (2.5)$$

Fifth, I estimated the relative (percent) fishing effort allocated to each gear (g) during each year (t) as

$$RE_{gt} = \frac{\hat{E}_{gt}}{\hat{E}_t} 100 \qquad (2.6)$$

Sixth, I estimated total fishing effort allocated to each gear by multiplying (i) proportion of effort allocated to that fishing gear by all fishers (RE_{gt}), (ii) mean fishing effort ($\bar{E}_t$), (iii) the population of participating villages (V_t), and (iv) the proportion of the population (P_t) who fished during a year (t) (adapted from (Teh and Sumaila 2013) to include effort and time)

$$E_{gt} = RE_{gt}\ \bar{E}_t\ V_t P_t \qquad (2.7)$$

Lacking other data, I assumed that the proportion of the population that fished (P_t) was static through time (Appendix C). Using proportions and estimates rather than raw sums of effort allowed us to compare across the study period, despite the temporally-varying sample sizes of fishers (Appendix C).

After obtaining estimates of total fishing effort, relative fishing effort, and the proportion of fishers using various fishing gears, I analyzed changes over time using generalized least square models (Zuur et al. 2009), Chapter 4). The explanatory variables were governance era and year. I assumed governance era accounted for changes in fishing regulations and governance structures. Year may indicate changes in pressures on the ecosystem, underlying changes in the abundance of species, or supporting ecological processes. Models incorporated temporal auto-correlation using a corARMA auto-correlation structure and used governance era as a variance covariate to allow for the heterogeneity of variances within governance eras (Zuur et al. 2009), Chapters 4 and 6). Due to high variance in the first decade, I restricted analyses of fishing activity to the period 1960-2010. I conducted all analyses in R 3.3.1 (R Core Team 2016) using packages dplyr, pgirmess, MuMin, and nlme.

2.4 Results

2.4.1 Eras of fisheries governance

In the past 60 years, fisheries governance in the Philippines has changed substantially. I split those changes into four eras, to which I have assigned periods and names: Limited Governance (1950–1971); Productivity (1972–1985); Decentralized Governance (1986–1997); and Co-management (1998–2010). These eras were distinguished by differences in governance priorities (e.g. production, sustainability) and relative influence of local vs. state institutions in the implementation of fishing policies (Table 2.3).

2.4.1.1 Limited Governance Era (1950–1971)

The Limited Governance Era was characterized by minimal government intervention. Fisheries policies were set at the national level, and little attention was paid to small-scale fisheries (Pomeroy and Carlos 1997, Batongbacal 2002). During this time, resources fell under state ownership and were open access, although local municipalities were able to grant licenses to commercial fishers in the first 5.5 km of coastal waters (BFAR 2004). Beginning in 1932, three destructive gears were banned nationally (blast, poison, electro–fishing), demonstrating some attention to conservation. The ban on these three types of destructive fishing remained in place throughout the period under study. At the local level, traditional management included the rights of some households to fish corrals, fish shelters, and mangrove trees (Guieb 2008).

2.4.1.2 Productivity Era (1972–1985)

In the Productivity Era, fisheries governance remained focused at the national level, with a series of legislation and development programs emphasizing increasing catches and maximizing resource extraction (Pomeroy and Carlos 1997, Batongbacal 2002). For example, in 1975 Presidential Decree 704 emphasized development, productivity, and building fisheries exports (Table 2.3). The decree paid little attention to conservation, but maintained existing gear bans and added restrictions on fine mesh nets. During this period, fishing subsidies allowed fishing effort and catches to increase (e.g. World Bank credits for motorized engines)(Pauly and Chua 1988, IDRC 2000). This focus on maximizing resource extraction was consistent with global practices at that time (Bailey et al. 1986). The first signs of overfishing and habitat declines appeared during the Productivity Era (Pauly and Chua 1988). Most records of emerging troubles, however, are from the Manila region and there is less documentation of trends in the central Philippines.

2.4.1.3 Decentralized Era (1986–1997)

The Decentralized Era, which began after the Marcos dictatorship lost power in 1986, saw the national government cut funding and devolve responsibility for management of natural resources – and other sectors (e.g. healthcare) – to local governments (Legaspi 2010). This structural transformation followed global trends in the 1980s and early 1990s, a time when decentralization was considered the best approach for economic development (Béné et al. 2010). The 1991 Local

Government Code formalized decentralization by shifting management of most natural resources from the national to the municipal level. As a result of this devolution of governance, municipal governments were given the responsibility of managing small-scale municipal fisheries (Batongbacal 2002). Furthermore, commercial fishing was prohibited within 15 km of coasts, though there was ongoing debate about how these boundaries were defined. There were no significant changes to fishing gear regulations during this Era.

2.4.1.4 Co-management Era (1998–present)

The Co-management Era began when sweeping fisheries legislation (Republic Act 8550:1998 Fisheries Code) strengthened the rights of small–scale fishers to exclusively use inshore waters up to 15 km from shore and established new institutional arrangements within the previously decentralized government. Small-scale fishers' rights to inshore waters were set both through moderately effective national regulations banning all commercial fisheries and through largely unenforced regulations restricting fishing to an individual's home municipality (Sotto et al. 2001). New institutional arrangements outlined in the fisheries code were designed to facilitate co–management. For example, the Fisheries Code explicitly required local participation by mandating that fisheries governance be shared among local governments, fishers' organizations, and NGOs. During the Co-management Era, some fisheries regulations were still set at the national level (e.g. some gear regulations) (Pomeroy and Carlos 1997, Calvan 2010). However, many aspects of small-scale fisheries governance, including the implementation of national and local policies, were conducted at the municipal level (Pomeroy and Carlos 1997, Batongbacal 2002, BFAR 2004, 2011). Existing research has found that, in some regions, the Co-management Era had stronger local capacity and participation than the Decentralization Era because local capacity and fisher involvement in management was gradually strengthened through the support of development programs and NGOs (Pomeroy et al. 2010, Ratner et al. 2012). However, other in areas fisheries management remained a low government priority (Béné et al. 2010). After the field work for this study was completed, the Philippines revised its Fisheries Code. The revisions (Republic Act 10654, 2015) contain a more explicit emphasis on ending illegal, unreported, and unregulated (IUU) fisheries and promoting conservation.

Under the 1998 Fisheries Code, municipalities are responsible for meeting nationally set targets for establishing MPAs to protect 15% of coastal waters (1998 Fisheries Code). Additionally, the country has committed to protecting 10% of reef areas by 2020 (Philippine Marine Sanctuary Strategy (2004)). In the Danajon Bank, the process of establishing MPAs has largely been driven by fishing communities rather than municipal governments (Ban et al. 2009b). To date, locally-enforced no-take MPAs have been established in over 40 locations (Alcala et al. 2008).

The Co-management Era, beginning in 1998, had several implications for fishing gears. First, all gear that damaged coral reefs, seagrass, mangroves or other marine life habitat became illegal. Second, active gears became illegal in municipal waters, although this restriction focused on gears deployed by boats (e.g. trawl, purse seines, drift gill net, tuna longline) that were deployed by boats. Other active gears (e.g. diving), however, remained permitted. Third, small-mesh nets (< 3 cm mesh) were prohibited in most cases, but remained legal for some species.

Table 2.3 A brief overview of four eras of Philippines fisheries governance.

Era	Years	Legislation	Major Components
Limited	1950[a] –1971	Admin. Code, 1917	National small-scale fishing regulations
			Allows municipal council to grant fishing rights
		Fisheries Act, 1932	Closed seasons, some access regulations
		Republic Act, 428	Limited commercial fishing inshore
			Three destructive methods banned (blast fishing, poison fishing, electro-fishing)
			Marine protected areas established at the municipal level
Productivity	1972–1986	Republic Act, 6451	National small-scale fishing regulation
		Fisheries Decree, 1975	Emphasis on development and productivity, with little attention to conservation
		Presidential Decree, 704	Ban on fine mesh nets, poison, blast, electro fishing
			Limit municipal boats < 3 gross tons
			Municipalities issue fishing licenses, but require national approval
			Municipal waters set at 3 nautical miles from shore
			MPAs set by national government
		Presidential Decree, 1219	Gathering coral banned
Decentralized	1986–1997	Local Gov. Code, 1991	Municipal fishing regulation of small-scale fisheries
			Prohibited commercial fishing within 15 km of shore (municipal waters)
Co-management	1998–2010[b]	Fisheries Code, 1998 (Republic Act, 8550)	Co-management of small-scale fisheries formalized
			Responsibilities shared by local institutions, stakeholders, NGOs, and national governments

Era	Years	Legislation	Major Components
			Destructive gears banned
			Some active gears restricted
			Marine protected areas established at the municipal level
			Target of protecting 15% of coastal waters
		Philippine Marine Sanctuary Strategy (2004)	Target of protecting 20% of coral reefs by 2020.

[a]The period under study began in 1950, but the fisheries laws in place at that time were from earlier legislation
[b]Following this study, the 1998 Fisheries Code was updated (RA 10654, 2015)

2.4.2 Indicators of co-management participation

When I examined proxies for governance participation during the Co-management Era, I found that 74% of villages had established fishers' organizations and 19% of fishers were members of fishers' organizations, with much variation in participation (0–80%) among villages. In all, 70% of study villages had locally implemented MPAs. Villages established MPAs between 1996 and 2007. All but one MPA were established after 1998, the first year that local governments had the autonomy to establish MPAs.

2.4.3 Trends in individual and aggregate fishing gears

During the four governance eras, fishers reported using 93 fishing gears, which I categorized into eight general classes: diving, nets, traps, hook-and-line, gleaning, corrals, poison, and blast fishing (Table 2.2; Appendix B). Some gears were designed to target specific species (e.g. jigs for octopuses) while other fishing methods aimed to catch anything of value (e.g. skin diving). Over their fishing careers, individuals used a mean of 2.47 ($\pm$ 0.07 SEM) gears (range: 1–7 gears over their careers). Individual fishers used anywhere from 1–6 fishing gears in a single year. The mean number of gears used by individual fishers in a given year increased over time (Kruskal-Wallis $X^2 = 20.94$ df = 3, $p < 0.001$), but remained below two in all eras (Figure 2.2a). Gear diversity (richness), the number of gears cumulatively used by all respondents, ranged from 9–75 gears per year. Rapid growth in diversity was evident from the 1950s onwards, peaking during the Productivity Era (Figure 2.2b; Kruskal-Wallis $X^2 = 21.62$, df = 3, $p < 0.001$). High Simpson's Index of Diversity was found across all eras, suggesting that most gears were used by a relatively small proportion of fishers (Figure 2.2c; range = 0.87–0.96, Kruskal-Wallis $X^2 = 14.27$, df = 3, $p = 0.003$).

2.4.4 Trends in individual and aggregate fishing

From 1960-2010 mean individual fishing effort remained fairly steady (range: 228-262 days per year; Figure 2.3a). In contrast, total fishing effort increased 2.4-fold, from approximately 576,000 fishing days per year in 1960 to approximately 1,363,000 days per year in 2010 (Figure 2.3b; Appendix C).

2.4.5 General fishing gears

I found oscillations in the dominant fishing gears for both common and rare gears (Figure 2.4). Initially, hook & line gears and nets had the highest total fishing effort, and this gradually transitioned to dominance by nets and diving.

- The total effort of nets increased 2.9-fold over time (Figure 2.4; Table D.1), while the proportion of fishing effort and the proportion of fishers using nets exhibited 1.2 and 1.3-fold increases, respectively (Figure 2.4; Tables D.2 and D.3). The

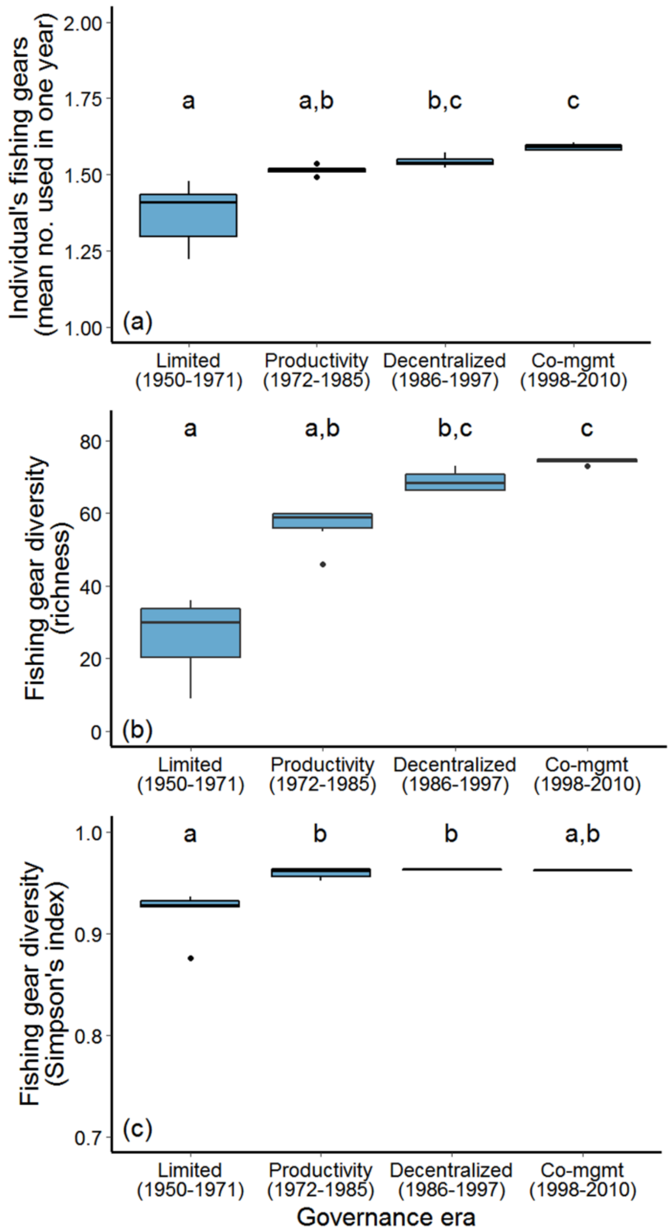

Figure 3.2 Changes in fishing gears during four eras of fisheries governance (1950-2010) (n=391 respondents). (a) Mean number of small-scale fishing gears used by individual fishers. (b) Richness of small-scale fishing gears (i.e. total number of gears used by all fishers). (c) Simpson's Index of Diversity of small-scale fishing gears used by all fishers. Fishing gears were classified as 93 specific gears and six randomly selected years were sampled during each Governance Era. Letters denote significant differences in gear use between Governance Eras at $p < 0.05$ as indicated by a Kruskal-Wallis Multiple Comparison post-hoc test. This change was largely due to the growing number of fishers.

greatest growth in the total effort and relative effort of nets occurred during the Productivity Era when nets exhibited a 1.5-fold increase in both metrics.

- Total and relative effort spent diving increased significantly over time, although increases varied among governance periods (Figure 2.4; Tables D.2 and D.3). Diving methods exhibited fairly steady total effort throughout the Limited and Productivity Eras, with fishing effort levels sitting at approximately 200,000 days per year (Table D.1). From the Decentralization Era through the Co-management Era (1986-2010) there was a 2.2-fold increase in total diving effort. The proportion of fishers using diving also increased over time, but at different rates during different governance eras. There was a strong increase in the proportion of fishers using diving throughout the Limited and Productivity Eras (2.5-fold increase). Since the mid-1980s, however, the proportion of fishers using diving has only increased slightly (1.1-fold increase).
- During the study period, both hook & line and trap gears exhibited moderately steady levels of total effort (1.4 and 1.1-fold increases, respectively), but their relative effort declined to 57% and 47% of initial levels (Figure 2.4; Tables D.1 and D.2). In particular, the relative effort and proportion of fishers using hook & line gears declined during the Productivity Era.

The four uncommon fishing gear categories showed highly variable patterns. In most years, these four gears each contributed to less than 6% of total fishing effort.

- Blast fishing initially comprised approximately 12% of total fishing effort. During the Limited Governance Era (1960-1971) blast fishing declined to a quarter of its initial levels, comprising less than 2% of total fishing effort by the early 1970s (Figure 2.4;

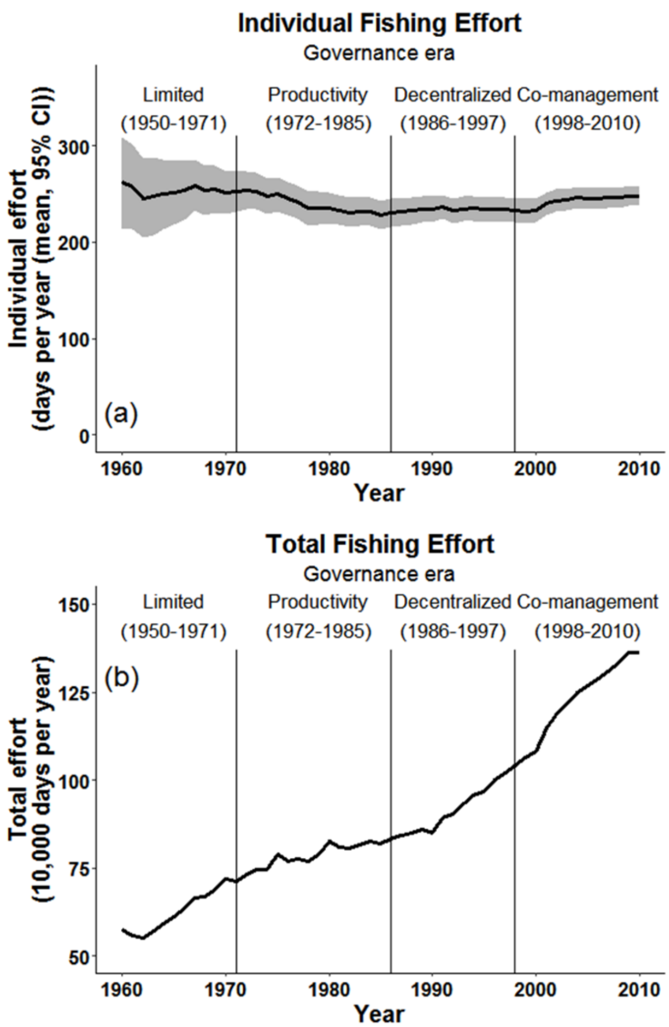

Figure 3.3 Long-term changes in fishing effort in the Danajon Bank, Philippines. (a) Mean individual fishing effort (95% CI). (b) Estimated total fishing effort (total number of fishing days by all fishers in 23 participating villages).

Table D.1). During the middle of the Productivity Era, the use of Blast fishing began slowly rising again – a pattern, which continued until the middle of the Co- management Era. In 2010 this highly damaging gear was used by a low share of all fishing effort (1.7%). However, this still amounted to more than 23,000 fishing days per year in 2010.

- Total effort by poison fishing was highly variable during early periods, and then exhibited a 1.8-fold rise in total effort under Co-management (Figure 2.4). In 2010 the total effort of poison fishing was similar to total effort of blast fishing.
- Total gleaning effort was initially variable and then increased 2.15-fold from the mid-1990s to 2010 (Figure 2.4). Currently, I estimate that total gleaning effort is over 40,000 days per year.
- Fish corrals were used with a relatively steady amount of total fishing effort (Figure 2.4; Table D.1). They increased 1.1-fold in total effort over time, and in 2010 were used approximately 44,000 days per year. In contrast, the relative effort of fish corrals decreased to 48% of their initial levels and the proportion of fishers using fish corrals decreased to 57% of initial levels (Tables D.2 and D.3).

2.4.6 Intensive, illegal and non-intensive fishing gears

The relative distribution of intensive and non-intensive categories was consistent over time (Figure 2.4). I report a steady rise in total effort by all intensive fishing gears over time. From 1960-2010 total fishing effort increased for destructive gears (3.4-fold increase), active gears (3.0-fold increase), and non-selective gears (1.5-fold increase). After the Decentralization Era began (1986 onward), the percent of fishing effort allocated to these three intensive gears and the proportion of fishers using these gears remained relatively steady. However, this steady pattern of relative fishing effort did not translate to total fishing effort. During the Decentralization and Co-management Eras, total fishing effort using these three intensive gear categories continued to increase.

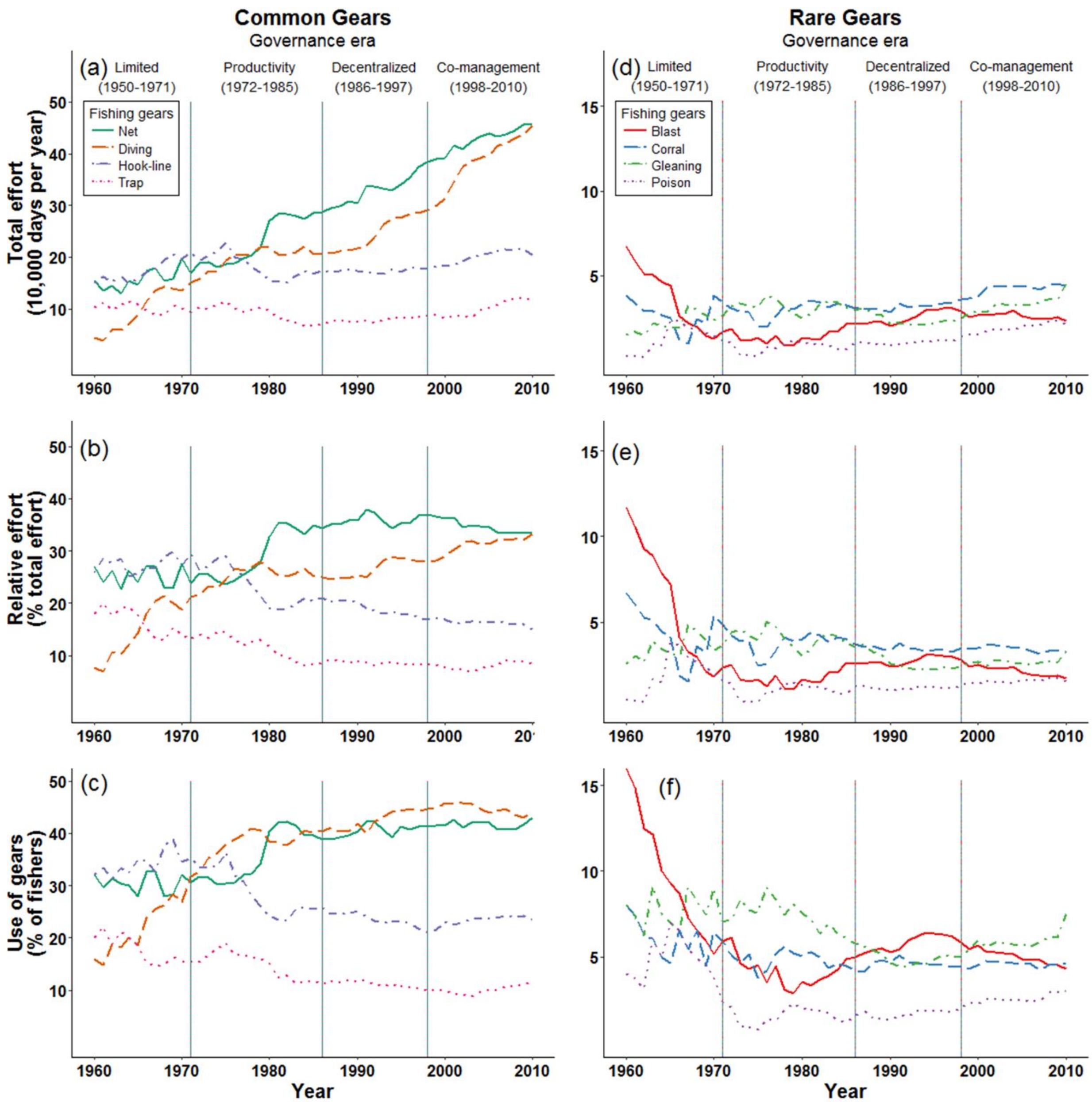

Figure 3.4 Long-term changes in fishing activities by multi-gear small-scale fisheries in the Danajon Bank, Philippines. (a-c) Changing use of the four most common fishing gear categories. (d-f) Changing use of four relatively rare fishing categories. (a,d) Estimates of total fishing effort by fishers from the 23 study villages. (b,e) Relative fishing effort. (c,f) Percent of fishers using these categories of fishing gears during any time in a year.

During our study period, total effort by illegal gears declined by a factor of 1.7 until 1998 (Figure 2.5; Table D.1). After the 1998 Fisheries Act, many existing gears became illegal. Thus the use of illegal fishing gears (total effort, relative effort, and the proportion of fishers) using illegal gears exhibited an 8.4 increase in 1998. Following this change, total effort by illegal gears remained relatively steady. However, in 2009 and 2010 total effort by illegal gears increased 1.2-fold from approximately 231,000 to 265,000 days per year.

Non-intensive gears also increased during the period under study, but these increases were more gradual than for their intensive counterparts. From 1960-2010 total fishing effort increased for non-destructive gears (2.2-fold increase), passive gears (2.0-fold increase), and selective gears (1.3-fold increase) (Figure 2.5; Table D.1). The proportion of fishers using non-intensive gears declined slightly over time, although these changes were relatively small (non-destructive and passive = 1.1-fold decrease; non-selective = 1.4-fold decrease) (Table D.3).

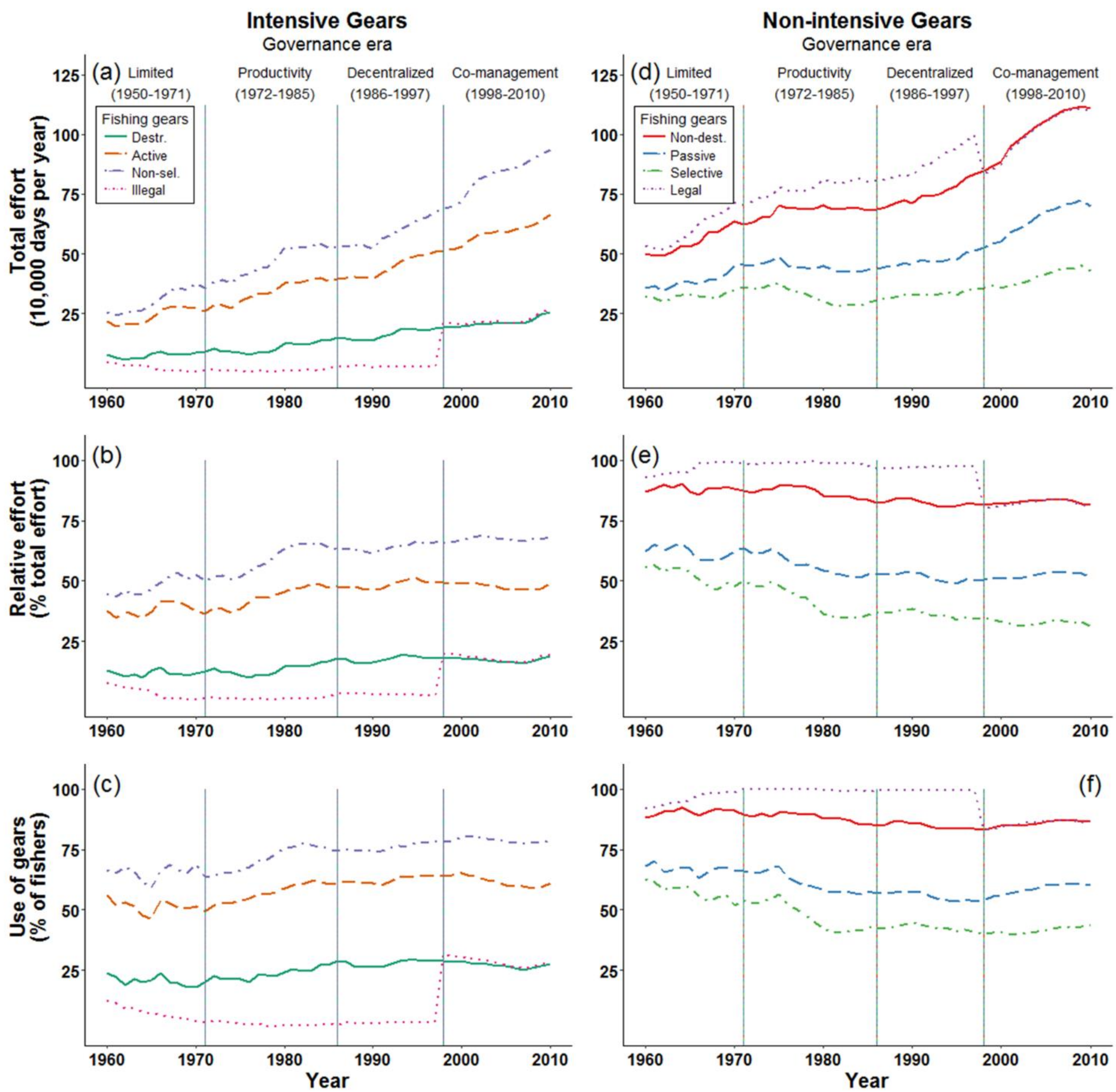

Figure 3.5 Long-term changes in fishing activities by multi-gear small-scale fisheries in the Danajon Bank, Philippines. (a-c) Changing use of four (non-exclusive) categories of intensive fishing gear. (d-f) Changing use of four (non-exclusive) categories of non-intensive fishing gears. (a,d) Estimates of total fishing effort by fishers from the 23 study villages. (b,e) Relative fishing effort. (c,f) Percent of fishers using these categories of fishing gears during any time in a year.

2.5 Discussion

Our rare and integrated analysis of small-scale fishing over six decades found that gears have become more diverse and that total fishing intensity and effort have greatly increased. All fishers targeting this ecosystem are using a growing diversity of fishing gears. Despite the systemic increase in cumulative gears used, most individual fishers used fewer than three gears over their entire careers and only one or two gears each year. I found large increases in the total effort by fishers using nets and diving, with other fishing gears showing less pronounced changes in total effort over time. When evaluating gears according to intensity, fishing intensified through increasing use of destructive, active, and non-selective gears. Mounting use of intensive gears outpaced the growth of non-intensive counterparts (i.e. non-destructive, passive, and selective gears). This burgeoning of gear diversity, intensity, and effort occurred before the mid-1980s, underscoring the value of a long-term perspective for understanding small-scale fisheries.

2.5.1 Benefits of gear diversification and persistence

This diversification of fishing gears may benefit fishers in four ways. First, targeting many species through gear diversification can improve the economic value of catches. For example, catching a large number of species benefited Kenyan fishers by increasing catch-per-unit-effort (CPUE) and mean trophic levels of catches (McClanahan et al. 2008). Second, new gears potentially enable catch of new species, a necessary substitution when original targets decline. In the central Philippines, for example, substitution allowed continuation of fishing after apparent extirpation of formerly targeted species (Lavides et al. 2010). Additionally, catching new species can allow fishers to take advantage of emerging opportunities, such as access to global markets for species such as seahorses, sea cucumbers, and abalone (Baum and Vincent 2005, Anderson et al. 2011). I infer that the shift I documented from dominant hook-and-line fishing effort to dive fishing effort indicates a corresponding shift in targeted species from primarily fishes (caught by hook-and-line (Kleiber et al. 2014)) to a mix of fish and invertebrates (caught by divers (Kleiber et al. 2014)). In the Danajon Bank, shifts in targeted species has been observed by previous research (Sotto et al. 2001), but not linked to corresponding changes in fishing gears. Third, new gears can provide a competitive edge for catching previously-targeted species in novel ways (e.g. new habitats; new life history stages). Fourth, modifications can improve gear efficiency and gear longevity. In European fleets, for example, small, step-wise modifications in gears during the last century and large changes in gears over the past two hundred years led to significant increases in catchability (Marchal et al. 2007, Thurstan and Roberts 2010). Here, respondents described how modifications of gears can improve gear longevity by incorporating durable materials (e.g. shifts from bamboo to plastic materials for traps). I infer that the increasing numbers of gears, combined with increasing distance travelled (Selgrath et al. 2017a), helped offset growing total fishing effort. In contrast, high fishing effort resulted in decreasing gear diversity in a Kenyan small-scale fishery where accelerated use of destructively efficient nets diminished the use of other gears (McClanahan et al. 2008).

I documented strong inertia of familiar fishing gears, highlighting the long-term implications of governance priorities and policies. I found the greatest relative increase in active and non-selective gears during a period when Filipino governance emphasized resource exploitation to meet economic goals including, putatively, food security (i.e. Productivity Era) (Bailey et al. 1986, Béné et al. 2010). During this period, development programs in the Philippines provided fishers with more efficient gears (Pauly and Chua 1988), which then lingered for the long-term.

In this case, the use of active and non-selective fishing gears persisted for decades, even after Filipino governance priorities shifted towards sustainable fishing (Green et al. 2004, Muallil et al. 2014). The 'stickiness' of these changes is evidence that policies created under the ethos of one era can have a lasting, and possibly unintended influence on future practices (Jentoft and Chuenpagdee 2009). Use of damaging gears may persist for many reasons including familiarity, poverty, social norms, lack of knowledge about the link between destructive gears and diminishing catches, and prohibitive start-up costs or time for learning new methods (McClanahan et al. 2005, Fabinyi 2007).

Where poverty is common, households frequently diversify their sources of revenue (Allison and Horemans 2006) often involving different livelihoods (e.g., gold panning, agriculture, forestry and fishing). It's conceivable that diversifying the methods used to maintain a single livelihood activity, such as fishing, could achieve outcomes similar to diverse livelihood portfolios. I found, individually, each fisher used relatively few gears, a choice with implications for their livelihood portfolios. Where livelihood opportunities are sparse, use of several gear types might arguably reduce a fisher's dependency on one group of species, as well as protect fishers from ecological (Jennings and Polunin 1996, Cinner et al. 2013) and market variability (Allison and Horemans 2006, Guieb 2008). Yet fishers on Danajon Bank did not adopt this strategy, instead relying on few gears (1-2) at any given time. This result supports previous findings that fishers may not perceive benefits from diversifying their gears (McCay 1981), or that fishers can be limited by the skills or capital investment required for new gears (Guieb 2008). The corollary is that fishers may perceive or acquire greater benefits by using existing gears (Béné and Tewfik 2001) or by diversifying into other livelihoods beyond fishing (Hill et al. 2012).

2.5.2 Quantifying historical fishing activities with LEK

A lasting ability to manage resources requires historical context for current conditions, relying on estimate of long-term trends. In data-poor situations, such estimates must rely on diverse data sources (Pauly and Zeller 2016). Our research confirms the role of local knowledge in developing a quantitative understanding of changes in resource extraction. Our work collected local knowledge in a technically rigorous fashion (e.g. through randomization of villages and respondents) providing information otherwise unavailable on six decades of fishing practices and their impact. I acknowledge that local knowledge has limitations, such as the loss of details over time. Thus, I assume that fishing estimates from the earlier years under study (e.g. Limited Governance Era) are less precise than later eras. This variance was reflected in the error measurements from that period. To some extent these limitations can be mitigated with appropriate sampling and survey designs (Means and Loftus 1991), such as those I used. Moreover, many of our approaches are flexible enough to be applied to many other resource management contexts.

2.5.3 Implications for fisheries management

Reducing fishing effort in small-scale fisheries requires striking a careful balance between ending overexploitation of ecosystems and building adaptive capacity within fishing communities. Our investigation of historical patterns is relevant to the challenges facing long-term fisheries management for a variety of reasons. Our results revealed large increases in total fishing effort by destructive, active, and non-selective fishing gears. This knowledge of past change supports establishment of appropriate fishing targets and planning for future changes a system can accommodate (Bavinck et al. 2005). For example, scaling back active and non-selective fishing methods to 1980 levels of effort would require a 56% reduction in fishing effort. Such reduction is similar to that recommended by other evaluations of small-scale fishing in the Philippines (Muallil et al. 2014).

Second, it will be both important and rather complicated to shift fishers' away from intensive fishing practices that may be well-entrenched. The "stickiness" of familiar fishing gears I observed also indicates that use of non-intensive gears persisted once established. The increasing prevalence of intensive fishing gears, however, combined with the large increases in total effort, potentially results in more ecological impacts than effort increases solely using non-intensive gears (McClanahan and Mangi 2001). Remarkably, there are few studies that focus on the effects of changing small-scale fishing gears in coral reefs, but existing evidence does point to benefits from eliminating intensive gear. For example, Kenyan fishing grounds that excluded beach seines – a destructive, active, and non-selective gear – had higher catches than sites where such intensive gears were used (McClanahan and Mangi 2001). Benefits were short-lived, however, later reduced by long-term increases in overall fishing pressure. Successful gear management could emphasize reductions in effort and intensive gears as synergistic goals. Any shift towards non-intensive gears would involve dual approaches encouraging existing fishers to switch gears and/or ensuring new fishers adopt less damaging gears. As the Philippines implements recent revisions to the 1998 Fisheries Code, prioritizing 'conservation, protection and sustained management of the country's fishery and aquatic resources' (BFAR 2015), managers should consider how to incentivize fishers to discard intensive gear and dissuade new fishers from their adoption.

Third (and a corollary of the second), managers should be vigilant regarding new emerging fishing gears. The progressive diversification of gears I observed occurred through both technology creep (modification of existing gears) (Marchal et al. 2007), as well as development of entirely new gears (Thurstan and Roberts 2010). Adaptions in net materials and mesh size substantially effect catches (Hicks and McClanahan 2012), but this facet of fishing gears was not consistently available from our interview data. A forward-looking advantage of the Philippines' broad fishing laws (e.g. prohibition of all habitat damaging gears) is that it encompasses new gears with undesirable externalities. Pro-active restrictions on development of new, intensive gears through regulations based on collateral impacts (e.g. habitat damage, low selectivity), rather than regulations focusing on specific gear types, could greatly aid fisheries management. Large marine protected areas may also provide refuges to offset the long-term environmental costs of damaging and non-selective gears (Thurstan and Roberts 2010).

2.5.4 Conclusions

Despite growing commitments to fostering sustainability in small-scale fisheries (FAO 2015), remarkably, little research has evaluated long-term changes in small-scale fisheries (Johnson et al. 2013). Our quantitative assessment of historical fishing practices targets this critical information gap and provides valuable advice for establishing sound methodologies to meet this goal. I determined that from 1950-2010, fisheries diversified, intensified, and increased in total fishing effort especially for intensive fishing gears. The priorities of fisheries governance were evident in the strong uptake of active and non-selective fishing gears during the Productivity Era. In many other cases, however, the fingerprint of governance was largely absent. Through this research, I demonstrate the fundamental role of fisheries policies and growing human populations in influencing changes in fishing effort as well as gears. This grounded historical understanding can provide direction and targets for ongoing efforts to reduce the effects of overfishing (McClenachan et al. 2012).

Chapter 3: Incorporating spatial dynamics greatly increases estimates of long-term fishing effort: A participatory mapping approach

3.1 Summary

The location and intensity of fishing is dynamic over time, thus greatly shaping ecosystems. In coral reefs, small-scale fisheries are frequently purported to cause declines in biodiversity and marine life populations. However, long-term information of fishing dynamics is often missing, making it difficult to assess this relationship. Within a marine biodiversity hotspot, I used participatory mapping to characterize long-term spatial dynamics of fishing to improve our understanding of the potential intensification and expansion of small-scale fishing. I evaluated fishing effort and gear use over 418 km^2 in the central Philippines during half a century (1960–2010), introducing a new method to estimate the sample size of fishers needed to accurately map the spatial extent of fishing. First, I compared aspatial and spatial estimates of total fishing effort (days per year), based on communities where interviews were conducted (50% of communities). Our aspatial estimate indicated at least a doubling over time, reaching 1.3 million fishing days per year in 2010. Spatial estimates showed fishing effort increased >18-fold at some locations, with the most concentrated effort in 1990. This 18-fold increase is twice that seen in spatially explicit estimates of global (industrial) fishing effort during the same period. Second, I evaluated how spatial characteristics of fishing had changed over time. The extent grew 50% by 2000, such that small-scale fisheries affected over 90% of the coastal ocean. The expanded area of fishing coincided with a greater spatial overlap among fishing gears, as well as a proliferation of intensive (destructive, active, non-selective) fishing gears. Use of intensive gears later declined

in some areas. The intensification and expansion of fishing shown here emphasize the need for spatial approaches to management that focus on intensive, and illegal, fishing gears. Such approaches are critical in targeting conservation actions (e.g. gear restrictions) in the most vulnerable areas.

3.2 Introduction

Quantifying fisheries' impacts and developing sound approaches to fisheries management requires understanding fishing effort in space and time (Walters 2003, Stewart et al. 2010). Since the mid-20th century, industrial fisheries have seen global increases in fishing effort (Anticamara et al. 2011), as well as a dramatic spatial expansion of fishing into new areas (Swartz et al. 2010) with corresponding impacts on marine life (Pauly and Zeller 2016). In industrial fisheries, these dynamics have catalyzed the uptake of technologies to track fishers through space and time (e.g. vessel monitoring systems (VMS)) (Lee et al. 2010), benefitting fisheries management. For example, spatial information enables correct interpret of catch-per-unit-effort (CPUE) trends (Walters 2003). Moreover, managers have leveraged spatial information about fishing to reduce overlap between high impact practices and sensitive marine life (Rosenberg 2000).

Small-scale fisheries also place significant pressure on the ocean, but knowledge of their spatial and temporal dynamics lags behind that of industrial fisheries. Small-scale fisheries are important for livelihoods and food security – particularly in the developing world (Teh and Sumaila 2013). The world's 22 million small-scale fishers have large impacts (Teh and Sumaila 2013), annually removing approximately 30% of global catches (22 million tonnes of marine life)(Pauly and Zeller 2016). Despite their importance, small-scale fisheries have been poorly documented and are often neglected by management (FAO 2015). Improving our understanding of spatio-temporal dynamics of small-scale fisheries could help inform recent international sustainability commitments (e.g., FAO Voluntary Guidelines for Securing Sustainable-Scale Fisheries) (FAO 2015).

A spatial and long-term perspective can provide a strong foundation for understanding how fisheries and ocean ecosystems interact. Historical decisions influence the structure and function of contemporary ecosystems (Tomscha and Gergel 2016), as well as the distribution of species and habitats (McKey et al. 2010). Historical information can help managers evaluate fishers' responses to past regulations (Walters and Martell 2004, p. 200). Historical perspectives provide context for how current conditions arose and the effectiveness of past governance (Ostrom 1990, Gelcich et al. 2010) enabling better decision-making (McClenachan et al. 2012). In coral reef systems, catch declines and ecosystem degradation are often attributed to a growing number of fishers, as well as the intensification of fishing (Mangi and Roberts 2006). Yet, little long-term data exist to contextualize changes seen in small-scale fisheries.

The dynamics of small-scale fisheries are challenging to document. Conventional fisheries data (e.g. catch records) are rarely available. Furthermore, many technologies developed for monitoring industrial fisheries are impractical. One major challenge is the sheer number of people involved (25 times more small-scale fishers than industrial). Additionally, small-scale fisheries tend to be decentralized (spread along a coastline) and occur in regions with limited funds, governance, and/or technical expertise to support advanced technologies (Johannes 1998, FAO 2015). As a result, less conventional approaches, such as use of local-environmental knowledge (LEK), can support small-scale fisheries management by providing the best estimates of fishing trends in data-poor situations (Hind 2015). The need to develop effective management is particularly relevant to small-scale fisheries targeting coral reefs in the Philippines; these reefs are a global conservation priority due to their combination of abundant biodiversity and heavy threats (Selig et al. 2014).

In this study, I quantify spatial and temporal changes in small-scale fishing over 50 years (1960–2010) and evaluate implications of these changes. Using a coral reef ecosystem in the central Philippines as a case study, I address three questions: First, do estimates of long-term fishing effort vary when using aspatial and spatial measures? Second, what are the spatial characteristics of fishing and how do they change over time? Third, how can maps quantifying historic fishing effort be used to strengthen conservation and management?

3.3 Methods

3.3.1 Study site

Our research focused on the central portion of the Danajon Bank ecosystem in the Central Visayas, Philippines (Figure 3.1; 10˚15'0'N, 124˚8'0'E). This double barrier reef supports variable conditions (e.g. turbidity, reef zones). The Danajon Bank sits off the coast of Bohol – a province with extreme poverty and minimal infrastructure (PPDO Bohol 2013). In the Danajon Bank, the period under study (1960-2010) was characterized by rapid human population growth (Philippines National Statisics Office 2010) and declines in catch-per-unit-effort (CPUE) (Christie et al. 2006). Fishing is a primary human activity influencing the Danajon Bank because

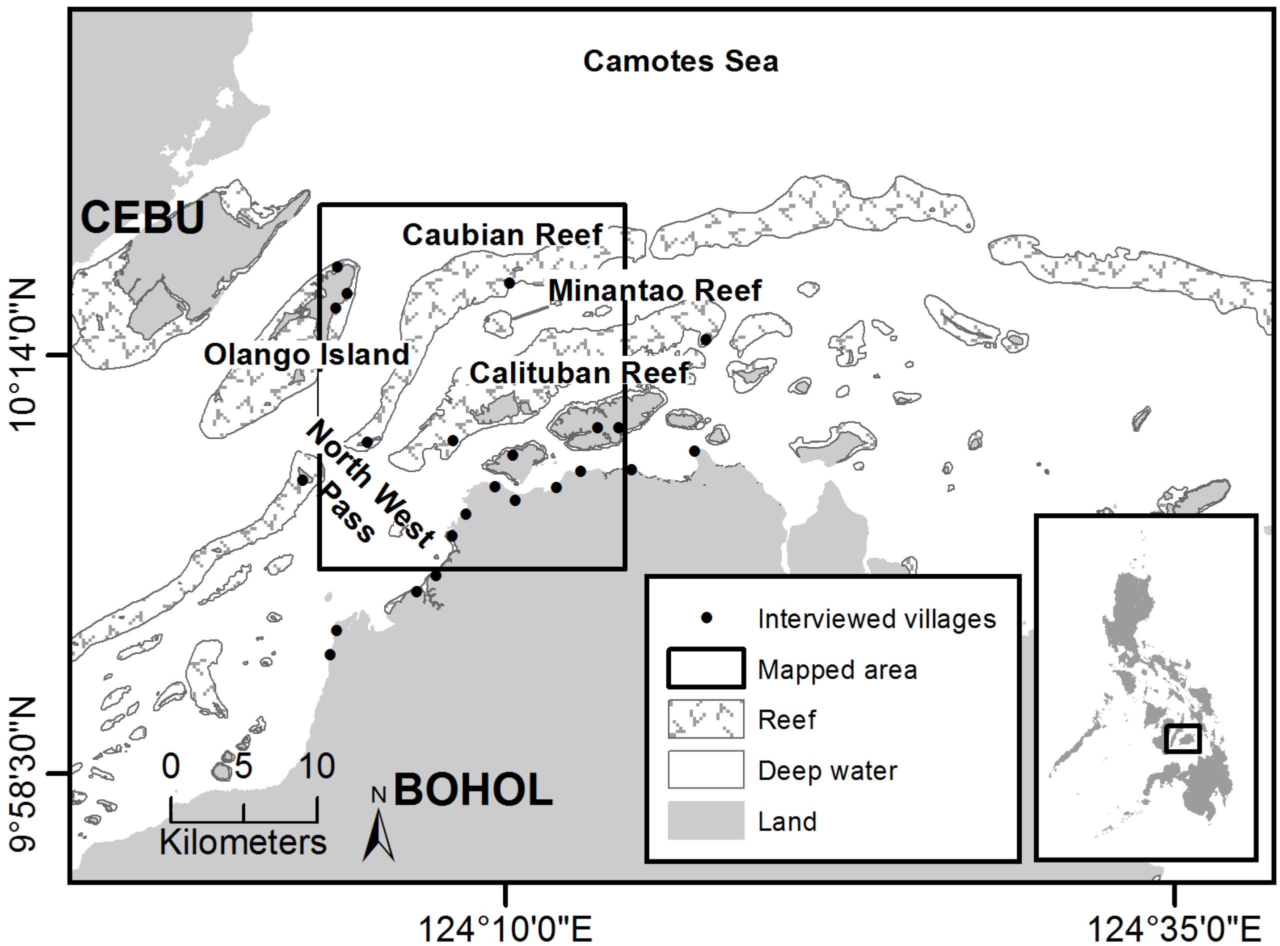

Figure 3.1 In the Danajon Bank Ecosystem (Philippines), respondents from interviewed villages were asked to map the history of their fishing practices inside of the mapped area.

of its high population density and lack of alternative livelihoods. The small-scale fisheries here are multi-gear and multi-species (Christie et al. 2006).

Fisheries management in the Philippines emphasizes co-management, gear restrictions, spatial restrictions, and marine protected areas (MPAs). In the Danajon Bank, these tools are implemented with varying degrees of effectiveness. For example, beginning in the 1930s, national gear-based restrictions banned blast and poison fishing with limited success. More recently the 1998 Fisheries Code (Republic Act (RA) 8550) established largely unenforced regulations restricting fishing to an individual's home municipality and restricting the use of all destructive, and some active and non-selective fishing gears.

3.3.2 Overview

Based on participatory mapping and interviews, I compared changes in long-term fishing effort obtained using both aspatial (days per year) and spatial estimates (days per year at location (i)), and then evaluated how spatial characteristics of the fishery had changed over time (1960-2010) (Figure 2). I consider how fishing effort has changed by comparing fishing effort during one year in 10 year intervals (e.g. cumulative days fished by all fishers in 1960, 1970, etc.).

3.3.3 Participatory mapping & fisher interviews

To document spatial and temporal changes in fishing in the central Danajon Bank, I conducted semi-structured, participatory mapping interviews in 2010 and 2011. During participatory mapping a resource user's historical experience and expert knowledge are used to create maps of local practices or environments (Chambers 1994). I interviewed 391 randomly selected fishers from 23 randomly selected villages and towns (Table 3.1, Figure 1). See Appendix A for interview questions. I sampled 50% of fishing villages within and up to 10 km from the mapped study area (Figure 1). This extended distance ensured our estimates of fishing effort accounted for those who fished in the mapped area, but lived elsewhere (Table 3.1). Prior to interviews, I obtained written consent from local mayors and oral consent from village officials and fishers.

Table 3.1 Fishers were sampled from 23 villages in the central Danajon Bank Ecosystem (Bohol and Cebu provinces, Central Visayas, Philippines).

Location	Municipality	Village†	Location	Distance to study area (km)	Village pop. (2010)	No. Fishers in village (2010)	No. Fishers interviewed (2010)	% fishers interviewed	% respondents fishing in study area
South-west of study area	Inabanga	Ondol	Rural Mainland	7	1098	131	9	7%	0%
	Inabanga	Lawis	Rural Mainland	5.4	2629	447	31	7%	16%
	Buena Vista	Asinan	Rural Mainland	2.8	757	71	5	7%	60%
	Buena Vista	Cruz	Rural Mainland	1.7	745	38	9	24%	56%
	Buena Vista	Cabulan West	Caye	1.2	1884	349	28	8%	82%
Inside of study area	Lapu Lapu	Baring	Ter. Island, Near City	n/a	2934	530	37	7%	100%
	Lapu Lapu	Talima	Ter. Island, Near City	n/a	4945	829	58	7%	100%
	Lapu Lapu	Tungasan	Ter. Island, Near City	n/a	1754	242	18	7%	100%
	Lapu Lapu	Caubian	Caye	n/a	2114	429	30	7%	100%
	Getafe	Pandanon	Caye	n/a	1833	320	21	7%	100%
	Getafe	Tugas	Rural Mainland	n/a	817	58	4	7%	100%
	Getafe	Nasingin	Caye	n/a	1441	335	25	7%	100%
	Getafe	Corte Baud	Rural Mainland	n/a	622	82	4	5%	100%
	Getafe	Poblacion	Rural Mainland	n/a	2210	36	3	8%	100%
	Getafe	Carlos P. Garcia	Rural Mainland	n/a	725	21	1	5%	100%
	Getafe	Jandayan Norte‡	Ter. Island, Rural	n/a	694	157	11	7%	100%
	Getafe	San Jose	Rural Mainland	n/a	1441	50	3	6%	100%
	Getafe	Tulang	Rural Mainland	n/a	1594	75	6	8%	100%
	Getafe	Mahanay	Ter. Island, Rural	n/a	448	21	2	10%	100%
	Talibon	Mahanay	Ter. Island, Rural	n/a	2012	113	8	7%	100%
East of study area	Talibon	Bagacay	Rural Mainland	0.5	3106	213	16	8%	94%
	Talibon	San Francisco	Rural Mainland	4.6	5870	408	25	6%	0%
	Talibon	Calituban	Caye	5.5	4527	552	37	7%	35%

† Villages are arranged from south-west to north-east.
‡ Number estimated from Getafe Coastal Resource Management Plan (2001 - 2005)

Data Sources: Village population = Village census data collected by village health centers; No. of fishers in village = Based on census data and our interviews with village health workers and other relevant village officials; % of fishers interviewed who are active in mapped area = (No. fishers interviewed who fished in the mapped area/ No. fishers interviewed) x 100

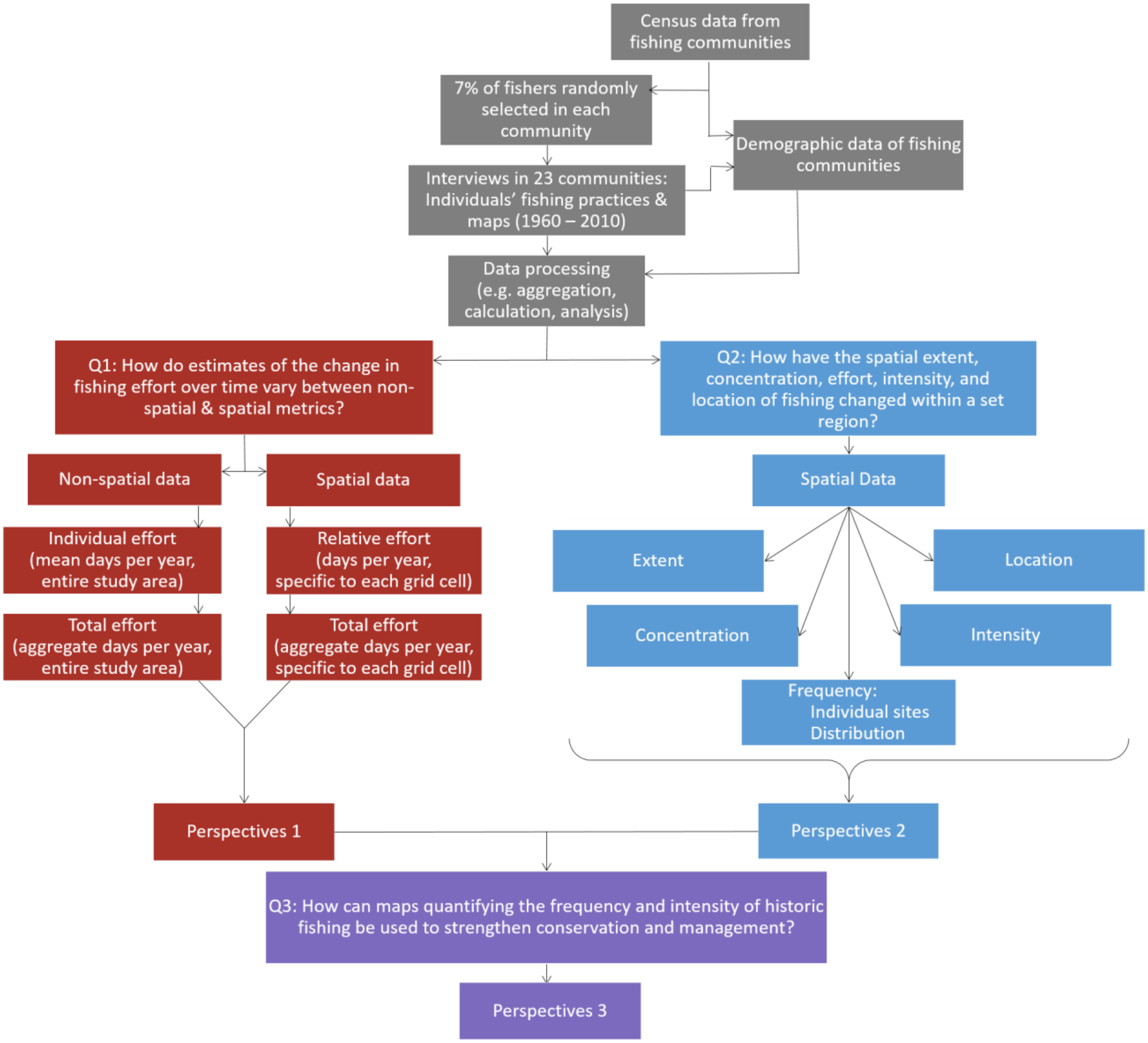

Figure 3.2 Schematic diagram depicting the relationship among data sources, research questions, and perspectives. At the top of the diagram, grey boxes show the relationships among data collection and processing that supported this research. Below, the boxes describe the data and estimates supporting Question 1 (red boxes) and Question 2 (blue boxes). Perspectives from Questions 1 and 2 were used to support Question 3 (purple boxes).

From the selected villages, I created a list of all fishers (full and part-time) using village census records. Census records did not include women who fished, and I did not attempt to correct this bias. From this list of fishers in each village, I randomly sampled 7% to interview (n = 391, with 295 of respondents fishing in study area). Since I was interested in a long time series, I stratified the fishers by age prior to random sampling. I focused our interviews on fishers who were born before 1981 because I estimated that most of this age group would have fished for at least 15 years. This method may underrepresent recent trends driven by young fishers, but prioritizes long time-series of fishing.

Interviews were conducted in the local language (Cebuano) by two local research assistants. Most interviews were conducted in respondent's houses or yards. Some interviews were private (only the respondent and the research assistant), while household members were present during other interviews. The setting was based on the respondent's preference. If a randomly selected fisher was unavailable, I either substituted other household members for the original respondent (i.e. if another household members also fished) or substituted the fisher with someone from a randomly selected backup list I made for this purpose.

After obtaining oral consent from a respondent, I used non-technical language to systematically build up information about the spatial history of his fishing practices (Hall and Close 2007). Interviews included six steps: personal history, fishing history, gear history, orienting fishers to the map, mapping his fishing grounds, and fishing ground history. Timelines went as far back as the respondents declared they could remember.

3.3.3.1 Documenting individual fishing histories

First, to improve the accuracy of recall dates in the interview process, I established a personal timeline for each fisher (e.g. year started fishing; year married), superimposed on a timeline of major events in the Philippines (see questions and timeline in Supplement S4). Second, I recorded years that the respondent fished in the Danajon Bank and any monthly/seasonal or migratory patterns in his fishing. From monthly effort patterns I calculated the total days per year a respondent fished in our study area. Several respondents migrated to fish, so I was careful to disentangle local from distant practices. Third, I made timelines for each fishing gear that a respondent used in our study area, recording the years fished with the gear. I used the respondent's fishing and gear history to confirm consistency with more specific questions about fishing grounds, later in the interview. Such triangulation helped internally validate interviews to ensure consistency (Chambers 1994, Neis et al. 1999).

3.3.3.2 Mapping fishing grounds

Fourth, I oriented respondents to the hardcopy base-map (later used to draw fishing grounds) by discussing and identifying various landmarks. I tested the respondent's understanding by asking him to locate places and describe map features until I was comfortable that the map made sense to the respondent. The base-map incorporated a high spatial resolution SPOT-5 satellite image (10 m by 10 m grid cell) to improve locational accuracy (Hall and Close 2007) and allow fishers to identify specific features in the landscape (e.g. the location of the reef crest). Additionally, the base-map identified anthropogenic landmarks, municipal centers, ports, and MPAs.

Fifth, I drew polygons around the respondent's current and past fishing grounds. The respondent directed us in drawing fishing grounds because many fishers seemed uncomfortable with drawing. Moreover, this method provided consistency among maps made by different fishers (e.g. minimum mapping units). Sixth, I made a spatially referenced timeline for each fishing ground mapped in the previous step (see timeline in Supplement S4). At each fishing ground, I recorded: years fished; months per year fished; days per month fished; and details about fishing gears for each year from 1960-2010. I recorded changes (often reported to be gradual) at intervals the fishers chose.

3.3.3.3 Estimates of fishing effort

When estimating an individual's yearly fishing effort, I used monthly patterns to estimate the days per year that a respondent fished. I took this approach because I felt it would be difficult for fishers to estimate directly how often they fished in a year, without building the number from smaller and more tangible sections of time. Additionally, I assumed that it was not possible for anyone to fish every day. Thus, during periods where fishers claimed to fish every day, I reduced their fishing effort to a maximum of 26 days per month. In many cases, fishers reported that their effort changed gradually, for example increasing from 5 to 10 days a month over a 5 year period. In this situation, I assumed that the change in effort was linear.

3.3.3.4 Fishing gears and engines

Fishers reported using 67 fishing gears in the study area (Appendix F). I created a database of fishing gears based on information derived from surveys, key informants, our own knowledge, and published literature. I evaluated changes in the total number of fishing gears used in a year (gear richness). I grouped gears into eight classes: blast fishing; fish corrals; gleaning; hook-and-line; nets; poison fishing; skin diving; and traps. I defined intensive fishing gears as those with a high magnitude of force, a notable efficiency, and/or a relatively low selectivity (Figure 3.3), recognizing their likely higher catchability (particularly for juveniles) and greater impacts (such as habitat damage). I further categorized intensive gears and their non-intensive counterparts into non-exclusive categories as follows: destructive/non-destructive (to reef habitats); active/passive; non-selective/selective; and illegal/legal (relative to when the gear was used). Fishing gears could belong to more than one category (e.g. trawls are destructive, active, and non-selective). Since the availability of boat engines is one factor influencing the distance that fishers travel, I also evaluated the percent of fishers who used engines in each year.

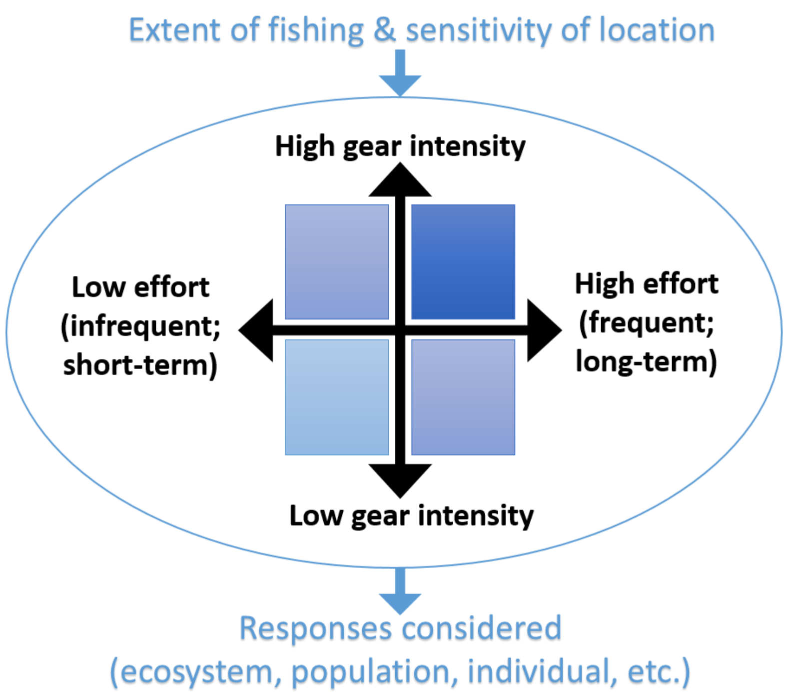

Figure 3.3 Factors influencing the ecological impacts of fishing.

3.3.4 Long-term changes in fishing effort

Next, I explored different metrics to characterize long-term fishing effort in aspatial and spatial ways for individual fishers, as well as cumulative measures for all fishers. I focused on fishing effort within in the study area. Since I sampled approximately 50% of fishing communities in the study area, our fishing effort metrics represent approximately half of the total fishing effort by all men who fished in the study area.

3.3.4.1 Aspatial fishing effort

Based on interview data, I quantified aspatial fishing effort using two metrics: (i) individual fishing effort and (ii) cumulative fishing effort by all fishers. This allowed us to evaluate two scales (individual vs. aggregate) over which effort may have changed.

First, I calculated mean individual fishing effort where $\bar{e}_{ft}$ is the mean of individual effort (days) (*e*) that respondents (*f*) fished during one year (*t*):

$$\bar{e}_{ft} = \frac{\sum_{1}^{f}(e_{1t} + e_{2t} \dots + e_{ft})}{f_t} \tag{3.1}$$

Second, I estimated the cumulative annual aspatial fishing effort where E_t is the total fishing effort (days) of all fishers from participating communities (*F*) during one year (*t*):

$$E_t = \bar{e}_{ft} \times F_t \tag{3.2}$$

3.3.4.2 Spatial fishing effort

To evaluate how fishing effort changed over time as well as space, I combined information from the fishing timelines and maps using five primary steps. (1) I digitized fishing grounds using heads up digitizing in ArcGIS 10.1 (Environmental Systems Research Institute, Redlands, California). Following digitizing, I standardized the widths of the fishing ground polygons that were located in deep areas. I decided this step was needed because deep fishing ground polygons had inconsistent widths (e.g. skinny vs. wide) based on which research assistant had done the interview. Standardizing was not necessary for shallow fishing ground polygons. In shallow areas, polygons were clearly aligned with the geographic features that were visible in the satellite images. I set the width of deep fishing grounds at 2/3 of the channel width, centered on the original polygon. After summing the maps, I found that there were some small gaps where no fishing was mapped between fishing grounds. As these were likely due to mapping errors, I smoothed maps by merging gaps < 10 ha with neighbouring polygons. Most gaps were small, with approximately 95% of gaps < 0.5 ha.

(2) I linked each respondent's fishing ground polygons with their reported fishing effort and gear history (based on interviews) to create maps of their total effort as well as gear-specific effort each year. I converted maps to raster format (20 m by 20 m grid cell), resulting in one fishing effort raster for each year a respondent fished, as well as additional maps specific to each gear in use that year. (3) For each year, I summed maps of individual respondents' to calculate cumulative fishing effort at each location where SpE_{fti} is the cumulative effort reported by all respondents (*f*) in year (*t*) in grid cell (i) and e_{fti} is the spatially explicit annual fishing effort for each respondent (*f*) in year (*t*) in grid cell (i):

$$SpE_{fti} = \sum_{1}^{f} e_{1ti} + e_{2ti} \dots + e_{fti} \tag{3.3}$$

Lastly, two additional metrics were created to enable more appropriate map comparisons over time. Maps of relative (proportional) fishing effort controlled for the different sample sizes in each year. Maps of cumulative fishing effort for all fishers (from participating communities) were extrapolated from maps of relative fishing effort using demographic information from the region (see Supplement S1 for details on demographic information. Cumulative fishing effort maps accounted for changes in sample size over time and allowed us to consider aggregate changes in effort by all fishers.

(4) For the relative effort map I converted the maps of absolute effort of respondents to maps of proportional effort, where *relative* SpE_{fti} is the proportional distribution of fishing effort by all respondents (f) in year (t) in grid cell (i) and E_{ft} is the cumulative number of days (E) that respondents (f) reported fishing in the study area in year (t):

$$relative\ SpE_{fti} = \frac{SpE_{fti}}{E_{ft}} \tag{3.4}$$

(5) I estimated the cumulative spatial fishing effort from communities where $cumulative\ SpE_{Fti}$ is the cumulative number of fishing days (E) by all fishers from study communities (F) in year (t) in grid cell (i). In this analysis, $\bar{e}_{ft}$ is the mean number of fishing days ($\bar{e}$) that respondents (f) fished in the study area in year (t) in grid cell (i):

$$cumulative\ SpE_{Fti} = relative\ SpE_{fti}\,\bar{e}_{ft}\,F_t \qquad (3.5)$$

I based the estimated number of fishers in year (F_t) on estimates of demographics changes (Table 3.2). Resulting maps were cropped to a uniform 19 x 22 km area to remove inconsistencies at map edges (total area = 418.0 km^2; ocean area = 354.1 km^2).

3.3.4.3 Additional information

I gathered demographic information from municipal offices, community census records, and during interviews. I was unable to obtain detailed information about village demographics in past decades. The Philippines census office aggregates data at the municipal level and does not distinguish between inland and coastal communities. Thus for past years, I calculated estimates of the fisher population size and the number of fishers targeting the study area using three assumptions: that village populations changed at the same rate as the rural population of the Philippines, that the proportion of village residents who fished was consistent over time, and that the proportion of people who fished in the mapped area was consistent over time. I based this estimate on the approach from Teh and Sumaila (2013) which estimates the number of fishers by multiplying an identified proportion of people who fish by the coastal population. I expanded this method for estimating the number of fishers back in time. I recognize that demographic changes differ among communities due to factors such as variable population growth, livelihood opportunities, and migration patterns. Since this detailed information was not available, our methods provide the best estimation of changing populations in fishing communities.

Table 3.2 Changes in the demographic characteristics of small-scale fishers and respondents from 23 villages in the central Danajon Bank Ecosystem (Central Visayas, Philippines).

Year	Estimated Fisher Population*	Estimated Fisher Population Targeting Study Area*	Age Of Respondents Targeting Mapped Area (Mean (SD))	No. Fishing Gears (Gear Diversity)	No. Respondents Fishing	No. Respondents Targeting Study Area	Estimated % Of Fishers Targeting Study Area Sampled In This Study	% Respondents Fishing In Study Area
1960	2199	1642	16 (5.0)	14	24	13	1%	54%
1970	2863	2138	20 (6.2)	23	79	41	2%	52%
1980	3502	2615	23 (8.3)	45	198	136	5%	69%
1990	3634	2714	28 (10.2)	55	302	206	8%	68%
2000	4633	3461	35 (11.5)	60	349	251	7%	72%
2010	5507	4114	45 (11.3)	60	374	249	6%	67%

Note: Estimates were calculated assuming that village populations changed at the same rate as the rural population of the Philippines, the proportion of people fishing remained the same, and that the proportion of fishers from various villages who fished in the focal area was steady over time.
* Estimated population is only from the villages that participated in this study. Those villages comprise approximately half of the villages in the area.

3.3.5 Spatial characteristics of fishing

To understand how the spatial characteristics of fishing activities changed within the region, I considered five dimensions of fishing: extent, effort, concentration, gear intensity, and location. Where necessary, I also controlled for sample size differences among years. Extent is simply the area (ha) fished within the map. To evaluate changes in effort, I assessed how cumulative fishing effort ($cumulative\ SpE_{Fti}$) changed in individual grid cells, and across the study area. The concentration of fishing was characterized by its level of spatial auto-correlation. Intensity of fishing gear was evaluated by analyzing maps from three subsets of gears: (i) all fishing gears; (ii) fishing gear categories (e.g. hook-and-line, nets); and (iii) four classifications of intensive fishing gears and their non-intensive counterpart: destructive/non-destructive; active/passive; non-selective/selective; and illegal/legal. Finally, I visually identified ecological and geomorphic characteristics of locations most targeted by fishing. (See Supplement S1 for descriptions). Focusing on relative effort ($proportional\ SpE_{fti}$) and total spatial fishing effort ($cumulative\ SpE_{Fti}$) allowed us to compare maps between years, despite different sample sizes in each year (Table 3.2). Analyses were performed using the programs ArcGIS 10.3 and R 3.2.4 (www.r-project.org).

3.3.5.1 Understanding spatial extent using area accumulation curves

For each year I quantified the area (km^2) that was fished. Additionally, I accounted for how differing sample size of fishers among years influenced the spatial extent of fishing. To do so, I adapted the rarefaction curve concept (Gotelli and Colwell 2001) to estimate the fisher sample size needed to map 90% of mean fishing extent in 2000 and 2010 (the years with the largest sample size in terms of fishers and area fished). In ArcGIS 10.3 I used bootstrapping to estimate the extent of fishing grounds mapped for each year with different samples sizes of respondents (range: 1 respondent – the maximum number of respondents in a year) (Payton et al. 2003). I used 10 iterations per sample size and removed two outlier fishing grounds.

3.3.5.2 Spatial patterns of fishing effort

I evaluated changes in the spatial patterns of fishing effort in two ways: via point pattern analysis, as well as grid-based analyses. I used point pattern analysis to assess how fishing effort changed over time in individual grid cells (i). I sampled 1000 randomly distributed points on the time series of maps. From these, I compared changes in fishing effort at site (i) between successive years (e.g. 1960 vs. 1970) using one-tailed paired sample t tests. Based on the statistical distribution of effort from all grid cells, I used two-sample Kolmogorov-Smirnov (K-S) tests (Mitchell 2005, p. 84) to evaluate changes in how much of the ocean was fished frequently or rarely. I also used the distribution of effort in all grid cells to assess the spatial autocorrelation of fishing effort, using Moran's I.

3.4 Results

During all decades, the mapped area was fished by people living both inside and outside of it (Table 3.1: Location). Of the 391 respondents, 75% fished in the study area during some or all of their fishing careers. As the distance increased from fishing communities to the study area, the percent of villagers who fished in the study area attenuated (Table 3.1: % respondents fishing in study area). In 2010, fishing villages showed large variability in the total number of fishers. They also exhibited high variability in the percentage of fishers who were active in the mapped area

(Table 3.1: % respondents fishing in study area). There was a 1.3-fold increase in the proportion of fishers who fished in the study area between 1960 and 1970. From 1980 onwards the proportion of people fishing in the study area stabilized (Table 3.2: % respondents fishing in study area). The mean percentage of fishers whose boats had engines was fairly steady from 1960-1990, then increased sharply in 2000 and 2010 (range: 17-47% boats with engines).

3.4.1 Demographic changes

As the distance increased from fishing communities to the study area, the percent of villagers who fished in the study area attenuated (Table 3.1: % respondents fishing in study area). In 2010, fishing villages showed large variability in the total number of fishers. They also exhibited high variability in the percentage of fishers who were active in the mapped area (Table 3.1: % respondents fishing in study area). There was a 1.3-fold increase in the proportion of fishers who fished in the study area between 1960 and 1970. From 1980 onwards the proportion of people fishing in the study area stabilized (Table 3.2: % respondents fishing in study area). Although the 1998 Fisheries Code restricted small-scale fishers to their home municipality (Batongbacal 2002), these seem to have little effect on actual travel.

3.4.2 Long-term changes in fishing effort

Individual fishing effort was consistent over time (mean: 218-254 days per year; Table 3.3: Individual fishing effort). There was, however, a large amount of variance in the number of days that individual respondents reported fishing in the study area. Some fishers fished year-round (full-time or part-time) while others fished in distant provinces, typically for eight months a year. Aspatial fishing effort – the total number of days that people fished inside the study area –

increased 2.5-fold from 1960 to 2010. By 2010, people from the study communities cumulatively spent over 1.3 million days fishing in the study area (Table 3.3: Aspatial fishing effort). If I assume that this estimate represents half of the cumulative effort (based on our random sample of 50% of fishing communities), I extrapolate that the study area supports 2.6 million fishing days per year by male fishers.

Mean total spatial fishing effort – the total number of days that people fished at specific sites (i) – peaked in 1990 at levels that were 21.6 times higher than 1960 levels. After 1990, mean spatial fishing effort slightly declined (Table 3.3: Spatial fishing effort; Figure 3.4b-c). Changes in both aspatial and spatial fishing effort were consistent with a substantial increase in the estimated number of fishers (Table 3.2: Estimated fisher population).

3.4.3 Spatial characteristics of fishing

3.4.3.1 Understanding spatial extent using area accumulation curves

During the fifty years under study, the spatial extent of fishing – within the study area – expanded by a factor of 1.6 (Table 3.3: Extent; Figure 3.4). The majority of the increase happened between 1960 and 2000, after which the extent of fishing remained steady until 2010 (Table 3.3: Extent; Figure 3.4).

The fishing area rarefaction curves (Figure 3.5) demonstrated that mapping 90% of the maximum fishing extent required a sample size of 125 fishers (rounded from n = 126). From 1980 onward there were adequate sample sizes (n >125 fishers). As fisher sample sizes in 1960 and 1970 were too small to accurately estimate the area fished (Table 3.2), these two decades

Table 3.3 Changes in the extent and effort (days per year) of small-scale fishing in the entire study area (aspatial) and at specific locations (grid cells; spatial) in the Danajon Bank Ecosystem (Philippines) (1960-2010).

		Individual Fishing Effort	Cumulative Fishing Effort: Aspatial†		Cumulative Fishing Effort: Spatial†		
Year	% Ocean Fished	Individual Effort (Mean (SD))	Days Per Year	Increase Since 1960 (Cumulative Effort)	Days Per Year (Max)†	Days Per Year (Mean (SD))	Increase Since 1960 (Mean Effort)
1960	59%*	241 (112)	529959	1	876	51 (126)	1
1970	71%*	254 (88)	727202	1.4	960	151 (205)	3
1980	79%	222 (107)	777444	1.5	2361	544 (543)	10.7
1990	85%	223 (109)	810382	1.5	5546	1104 (1102)	21.6
2000	92%	218 (111)	1009994	1.9	5178	1049 (1057)	20.6
2010	92%	244 (94)	1343708	2.5	4213	924 (790)	18.1

* Fishing extent estimated based on modelling.
† Estimations based on estimated number of fishers from study villages and the mean number of days fished by respondents in that year.

required that estimates of fishing extent be based on modeling. I compared linear and quadratic models with observed extents from 1980-2010. The quadratic model of fishing area had the best fit (quadratic $r^2 = 0.98$; linear $r^2 = 0.91$). Thus, I used fishing extent estimates for 1960 and 1970 derived from the quadratic model (Figure 3.5).

3.4.3.2 Fishing effort: All fishing gears

In our study area, the diversity (richness) of fishing gears used in a year increased over time from 14 gears in 1960 to 60 gears in 2010 (Table 3.2). When I compared spatial fishing effort at randomly distributed points through time, I found that effort increased significantly in individual grid cells (i) over the first three time-steps (t test: $p < 0.001$; Table E.1). After the year 2000, spatial fishing effort at individual grid cells (i) did not change significantly (Table E.1). When I assessed how much of the ocean was fished frequently or rarely, there were significant changes in each time step (Figure 3c; K-S test: $p < 0.001$; Table E.3). From 1960-2000 there was a shift from (i) no sites with high levels of fishing to (ii) several sites with high levels of fishing. In the year 2010, this pattern began to reverse as the spatial distribution of fishing effort became somewhat less concentrated. Over the 50 year period under study, fishing remained highly concentrated (Moran's I > 0.96 for all years and all gear categories).

3.4.3.3 Fishing extent and effort: Categories of fishing gear

The extent of the four most commonly used fishing gear categories (hook-and-line, nets, diving, traps) expanded over time (Figure 3.6; Table E.2) and the spatial distribution of fishing effort for these four gears changed significantly in each decadal period (K-S test: $p < 0.001$; Table 3.1). In 1960, hook-and-line fishing was the predominant fishing gear and was used

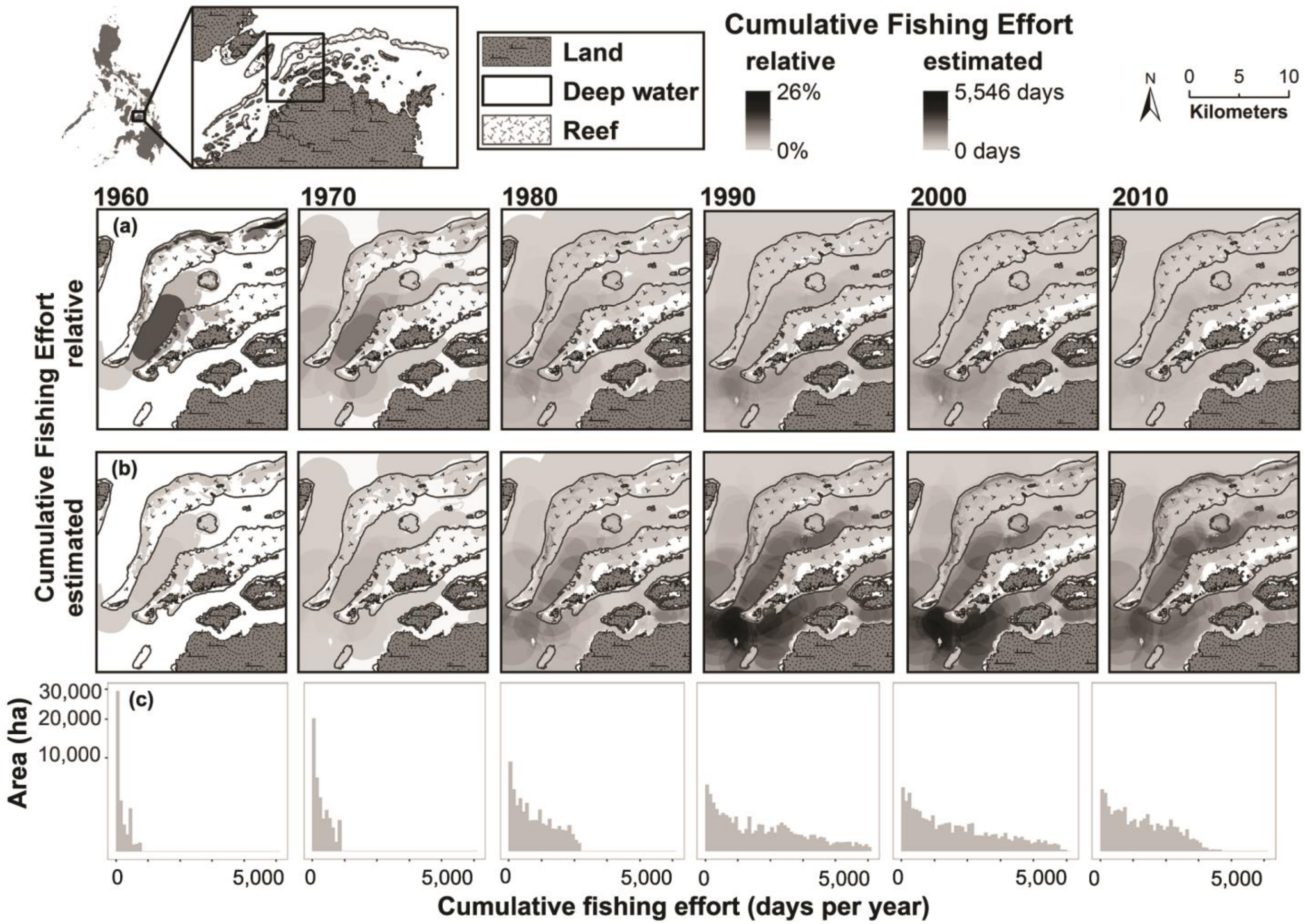

Figure 3.4 Spatial fishing effort (days per year) in the Danajon Bank Ecosystem (Philippines) from 1960-2010. (a) Relative maps show the percent of the total fishing effort. (b) Estimated maps show the cumulative fishing effort by fishers from interviewed villages. (c) Histograms show the area (ha) affected by varying levels of fishing effort. Land areas were excluded from the analysis.

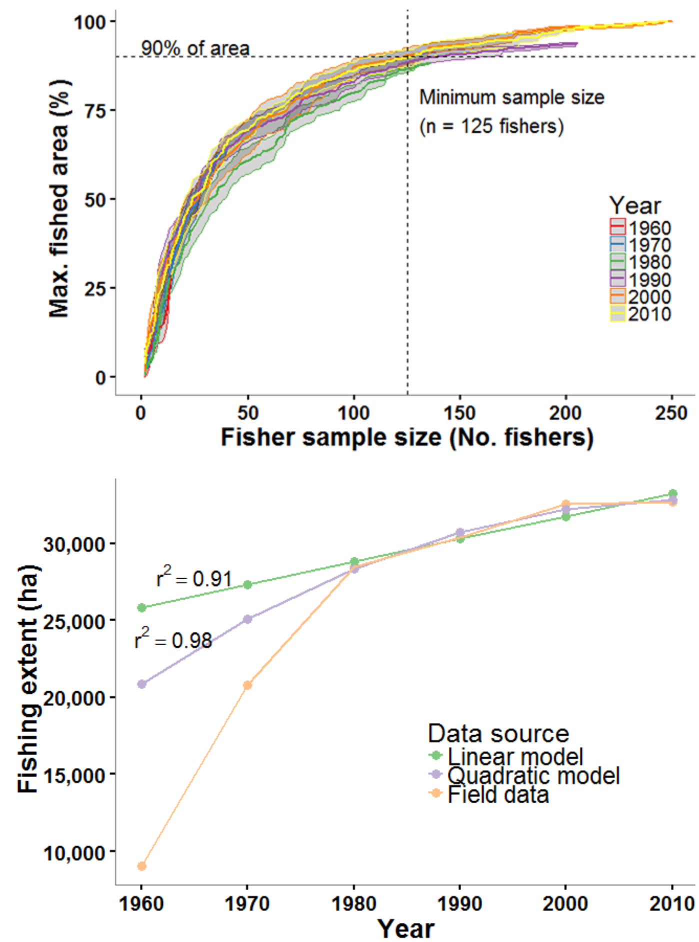

Figure 3.5 Estimations of fishing extent in the Danajon Bank Ecosystem (Central Visayas, Philippines) from 1960 – 2010. (a) Rarefaction curves for the area-fisher relationship, or the number of fishers needed to sample the spatial extent that is fished. Based on samples from 2000 and 2010 (the years with the largest sample size), a sample size of 125 fishers is needed to sample 90% of the fished area. (b) Model-based estimates of fishing extent.

in approximately 10% of the study area. At the same time nets, diving, and traps were used in less than 5% of the study area (Figure 3.6). From 1970—2010, nets' extent doubled, which was the greatest areal increase for any fishing method. In 2010 hook-and-line fishing and nets were the two most widely used fishing gears (maximum extent = 71%, Figure 3.6). From 1990–2010, blast fishing was used in approximately 20% of the ocean, while poison fishing was more limited in extent (area = 2%; Table E.1; Figure 3.7). During the same time, the area gleaned by fishers more than doubled from 3% to 7%, while fish corrals remained scarce (< 0.05% of the ocean; Table E.3; Figure 3.7). When comparing fishing effort at individual sites, cumulative fishing days for hook-and-line and nets grew over time (t test: $p < 0.001$), peaking in 1990 (Table E.1). Dive fishing was the only category of fishing gear in which the number of fishing days at individual sites increased significantly during every decade (t test: $p < 0.001$; Table E.1).

3.4.3.4 Fishing extent and effort: Intensive fishing gears

I classified 18% of gears as destructive, 49% of gears as active, and 68% of gears as non-selective. Many gears were illegal (20%), with 15% of all gears becoming illegal after changes in fisheries regulations (1998 Fisheries Code). During the five decades under study, intensive fishing gears were widely used throughout the study area (Table E.1; Figure 3.8). At individual sites, the cumulative number of days fished for all four categories of intensive gears increased from 1970 to 2000, but did not change significantly from 2000 to 2010 (Table 3.1).

From 1960 to 2010 there was a 5.3-fold increase in the spatial extent of destructive gears (Table E.1; Figure 3.8a) and mean fishing effort by destructive gears grew by a factor of 39 (Table E.1). Over time, the spatial extent of active gears grew 4.4-fold to target 88% of the mapped ocean (Table E.1; Figure 3.8c). Non-selective gears consistently were used in a broader area than selective gears (Table E.1; Figure 3.8e). Over time, mean cumulative days per year of active and non-selective fishing effort increased by a factor of 14. There was a three-fold increase in the extent of illegal fishing from 1990 to 2000 (Appendix

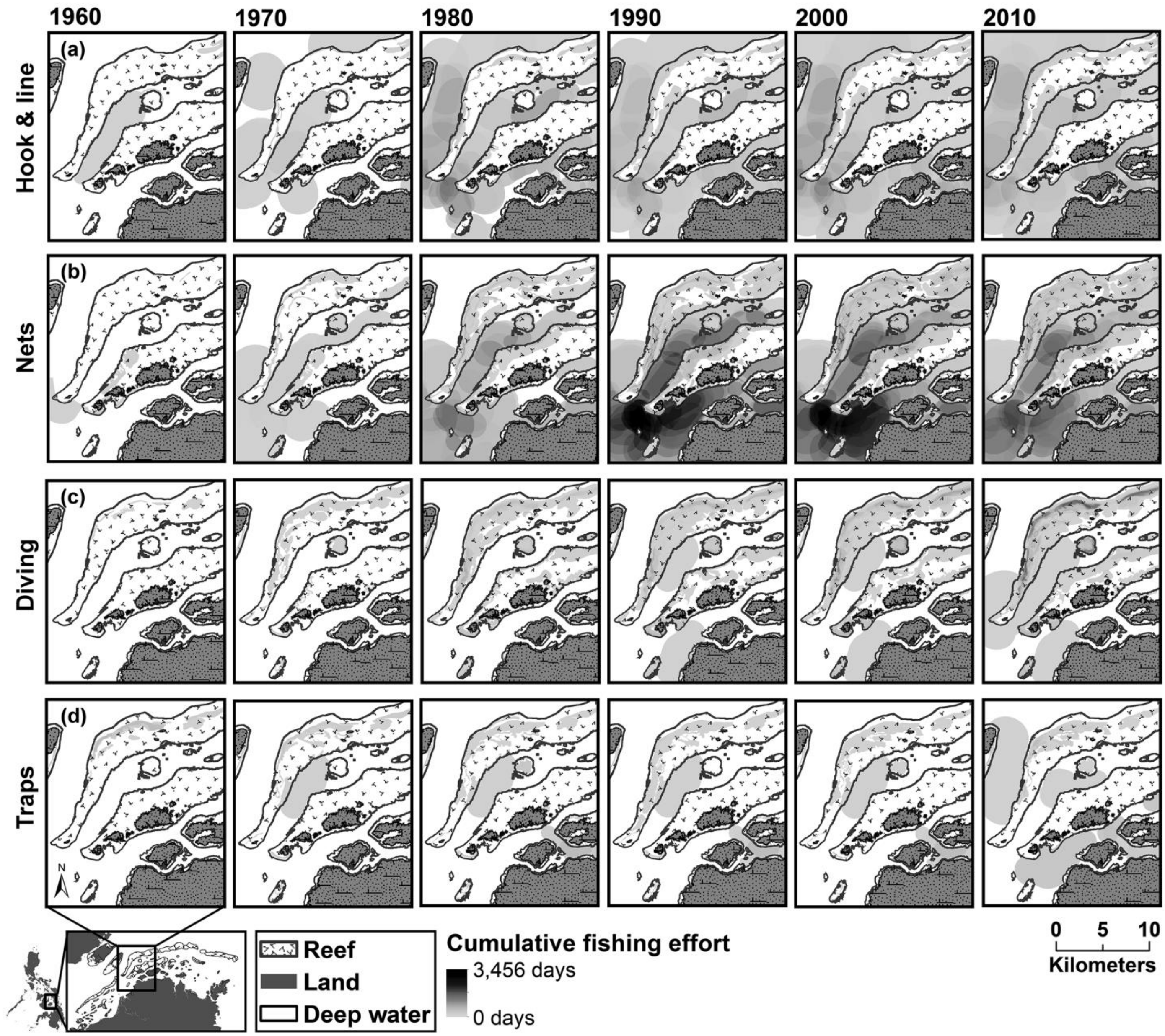

Figure 3.6 Spatial fishing effort (days per year) in the Danajon Bank Ecosystem (Philippines) from 1960-2010 for the four most commonly used classes of fishing gears. Colors are comparable among years and among gears.

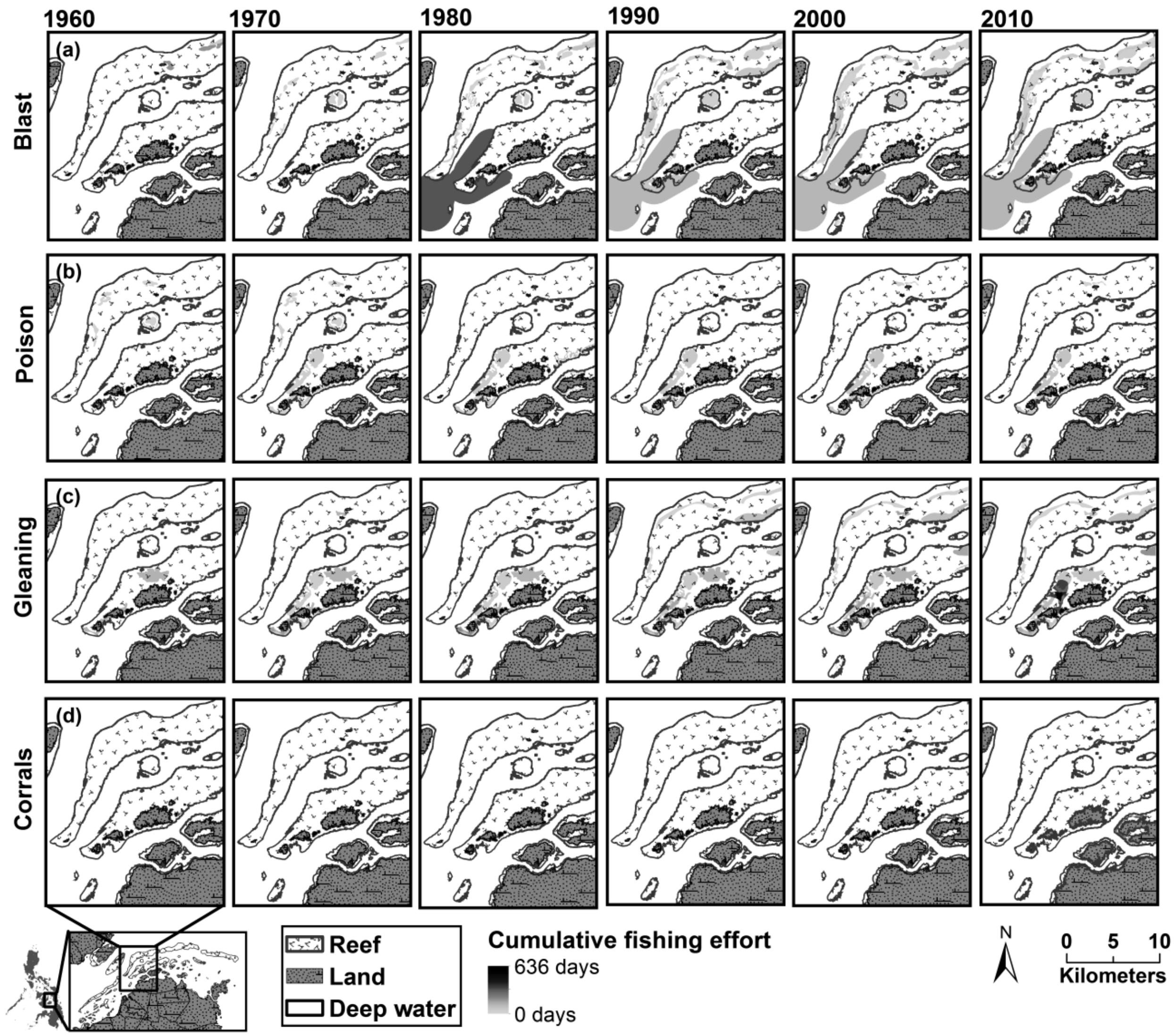

Figure 3.7 Spatial fishing effort (days per year) in the Danajon Bank Ecosystem (Philippines) from 1960-2010 for the four relatively rare classes of fishing gears. Colors are comparable among years and among gears. Note that only gleaning was used more than 375 days per year in a single location (2010).

Table E.1; Figure 3.8g). Mean total fishing effort of illegal fishing increased by a factor of 48. The major increase in the extent and effort of illegal fishing gears occurred when all destructive gears, many active gears, and most small-meshed nets became illegal under the 1998 Fisheries Code.

3.4.3.5 Locations of fishing effort

Overall, the most heavily fished areas were located in channels, with the highest concentration of fishing – including destructive fishing – located in the Northwest Pass (Figures 3 and 5; see Figure 1 for site names). Through time there was a gradual increase in cumulative spatial fishing effort for all fishing at the northern slope of the Caubian Reef (Figure 3.3b). This location is isolated from most villages (Figure 3.1). The trend of growing fishing effort on the northern Caubian reef became more pronounced in 2010 (Figure 3.3b). This trend was associated with an increase in destructive gears on some reef slopes (Figure 3.8a), as more fishers began using destructive diving methods to catch invertebrates.

Various fishing gears initially were used to target distinct areas in the ecosystem (e.g. reef flats; deep channels). As the use of fishing gears spatially expanded, however, so did their overlapping distribution (Figure 3.6). Through 1980, hook-and-line fishers predominantly used deep areas, but over time they began fishing more often in shallow reef areas (Figure 3.6a). Net fishers targeted both deep and shallow areas, but did not use nets in the deeper and more exposed Camotes Sea (Figure 3.6b). Respondents who used diving primarily fished on reef slopes at offshore reefs (Figure 3.6c). This pattern was consistent over time, but from 1980 diver density began increasing in reef flats and reef slopes of the inshore reef. Trap fishers initially targeted shallow areas, but began fishing in deeper areas from 2000 (Figure 3.6d).

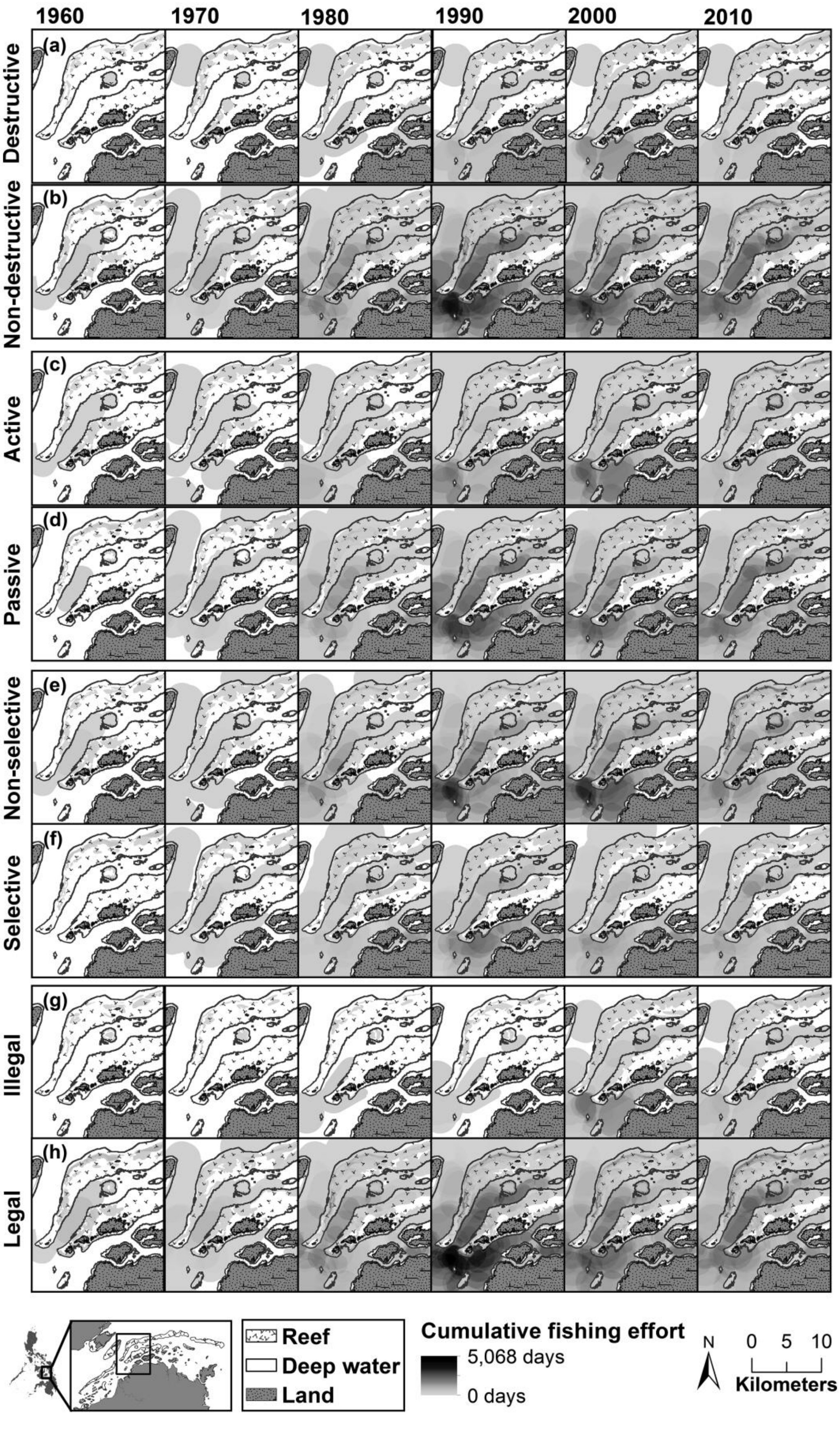
1960
1970
1980
1990
2000
2010
(a)
Destructive
(b)
Non-destructive
(c)
Active
(d)
Passive
(e)
Non-selective
(f)
Selective
(g)
Illegal
(h)
Legal
Reef
Deep water
Land
Cumulative fishing effort
5,068 days
0 days
N
0 5 10
Kilometers

Figure 3.8 Spatial fishing effort (days per year) in the Danajon Bank Ecosystem (Philippines) from 1960-2010 for intensive and non-intensive categories of fishing gears. Colors are comparable among years and among gears.

3.5 Discussion

My fifty-year analysis of small-scale fisheries in the Danajon Bank offers a rare in-depth look at the spatial and temporal development of small-scale fishing, one of the major influences on coral reef ecosystems (Johnson et al. 2013). When comparing spatial and aspatial metrics, I found that ignoring space greatly under-estimated the escalation fishing effort (aspatial: 250% vs. spatial: 1,800% increase from 1960-2010). Here I quantified five mechanisms potentially affecting these fisheries: (1) a rapid increase in cumulative fishing effort (days per year) – driven by significant growth in the number of fishers; (2) an expansion of fished areas; (3) a diversification of fishing gears; (4) a proliferation of intensive fishing gears; and (5) a growing overlap among multiple types of fishing gears. Further, I developed a novel method – area accumulation curves – to estimate the sample size of fishers needed to accurately estimate the entire fished area. My methods are highly transferrable to other data-poor small-scale fisheries impacted by growing fishing pressures.

3.5.1 Mechanisms of change: increasing effort, spatial expansion, evolving fishing gears

The sharp increase in fishing effort I illustrated in this case study remain an enduring challenge for small-scale fisheries management in the Philippines and elsewhere (Pauly and Chua 1988, Muallil et al. 2014). Increasing effort can increase annual catches up to a point, but also creates higher variability and produce lower trophic level catches (McClanahan et al. 2008). Significant tension exists between the recognition that the current number of fishers is unsustainable, and the seemingly insurmountable task of reducing the number of fishers. Impediments to lowering fishing effort include a growing human population (www.nscb.gov.ph/secstat/d_popn.asp), a lack of institutions that limit entry to the fishery (Ostrom 1990), few alternatives livelihoods (Hill et al. 2012), and feedbacks between poverty and overexploitation (Muallil et al. 2014). Since the large number of fishers drove the incredibly high levels of fishing effort, it is unlikely that this small-scale fishery – and others like it (Muallil et al. 2014) – are sustainable unless fewer fishers are involved. I base this conclusion on our documented fairly stable level of individual-level fishing effort and the moderately steady relative fishing effort. These unwavering trends of individual effort are in sharp contrast to the large increase in the total number of fishers.

Spatial expansion of fishing, such as I documented here, can influence the distribution of fishing effort and can occur in response to catch declines or new technologies (Walters 2003, Daw 2008). Similar to Brazilian fisheries, I documented that locations of the most heavily fished locations were moderately stable over several decades (Begossi 2006). Unlike Brazilian fisheries, however, I report a simultaneous expansion into previously unfished areas. Our respondents discussed conditions driving their spatial expansion, including growing competition and the need to use less desirable fishing grounds after preferred spots were degraded. These reasons were consistent with the growing number of fishers documented here and widespread reports of environmental degradation (e.g. Marcus *et al.*, 2007). An additional factor allowing fishing to expand and shift was likely the growing number of engines, particularly from 2000-2010. During these decades more engines corresponded to increasing effort on the relatively remote northern reefs and, unexpectedly, to a decrease in the spatial fishing effort at the most concentrated sites. I hypothesize, that the growing availability of engines allowed some fishers to transfer their effort from heavily fished locations to areas beyond our study site as recently seen in Nicaragua (Daw 2008) and over the past two hundred years in Scotland (Thurstan and Roberts 2010).

The type of fishing gears used influences the environmental impacts of fishing and intensive gears amplify the impacts of high fishing effort (Mangi and Roberts 2006, Thurstan and Roberts 2010). Despite this knowledge, our work represents one of few attempts to quantify changes in small-scale fishing gears over time (Johnson et al. 2013). A small, but growing body of work on small-scale fishing gears has included estimates of how many people use gears, collateral impacts, and catches (e.g. (Saenz-Arroyo et al. 2005, Begossi 2006, Hicks and McClanahan 2012). I provide a different perspective by quantifying aggregate changes in the gears used in an ecosystem over five decades. Reconstructing these changes has been possible for fisheries that were documented in some way (e.g. Scottish fisheries) (Thurstan and Roberts 2010), but the historical transformation of fishing gears has not previously been quantified for small-scale fishing. I report three specific developments: the increasing diversity (richness) of fishing gears, the increasing intensity of fishing gears, and the increasing extent – and therefore overlap – of different fishing gears.

Having identified such specific changes, I can identify management targets and research gaps. For example, the range expansion of fishing and concurrent use of multiple different gears has unknown, yet likely important, effects on fishers and reef systems. Elsewhere, changes in fishing practices have emerged out of necessity (e.g. species substitutions when original targets decline) and opportunity (e.g. emerging technologies or global market access) (Saenz-Arroyo et al. 2005, Lavides et al. 2010, Anderson et al. 2011). However, the ecological effects are less clear. It is unknown how growth in gear overlap will interact with other stressors to influence coral reefs, particularly at high levels of total fishing effort (Ban et al. 2014).

3.5.2 Historical participatory mapping can strengthen conservation & management

When retrospective participatory mapping is designed to account for biases, as with the design of this study, this approach can provide valuable information about past practices. Historical LEK is influenced by the presence of recall inaccuracies, the tendency for past memories to be influenced by windfall events (e.g. unusually large catches), and the under-reporting of illegal practices (Gavin et al. 2010, O'Donnell et al. 2012). I thus assume that data from earlier years (1960, 1970) are less precise than later years. I aimed to minimize biases by adopting a technically rigorous approach including randomization of communities and respondents, and internal validation of responses (i.e. triangulation) (Chambers 1994). Snowball sampling has been promoted in fisheries research, but during a pilot study I found that this method led to two significant biases: under-sampling of illegal fishing gears and over-sampling of fishing gears used by our initial respondents. I also aimed to improve accuracy by focusing on fishing activities that often changed gradually (e.g. gear use and effort). These activities can be recalled more accurately than those with large fluctuations, such as catches (Neis et al. 1999).

Historical maps can lead to better-informed policies in data-poor systems. Maps of historical fishing can help evaluate the influence of past governance approaches (e.g. 1998 Fisheries Code) and be used to set future conservation targets. For example, one could build on our mapping methods to set conservation priorities in locations where vulnerable species and habitats coincide with mounting fishing pressure (Soykan et al. 2014). Furthermore, because it is appreciated that effective conservation hinges on local support, participatory mapping can enhance effectiveness by creating a space for local engagement and trust (Chambers 1994). Participatory mapping can also improve local buy-in by providing opportunities to identify options that meet conservation targets and minimize impacts to fishers (Klein et al. 2008).

3.5.3 Conclusion

Our quantitative assessment sheds light on the significant transformations of small-scale fisheries over the past half century. I demonstrate participatory mapping of long-term fishing can foster a deeper understanding of otherwise poorly documented fisheries and can be used to contextualize the conditions found in today's oceans (McClenachan et al. 2012). From this approach, I identified five mechanisms through which fishing has changed: effort, extent, diversity, intensity, and overlap. Armed with these mechanisms – and supported by 2015 revision to the Philippine Fisheries Code (RA 10654) – managers can work to scale back fishing activity in this overfished ecosystem. However, outcomes will be limited by community support and will depend on complementary strategies to address the poverty and overpopulation underlying fishing transformations (D'Agnes et al. 2010). Where stakeholder buy-in is established, maps of historic fishing effort can allow managers to set achievable targets for fostering sustainability in small-scale fisheries.

Chapter 4: Mapping for Coral Reef Conservation: Comparing the Value of Participatory and Remote Sensing Approaches

4.1 Introduction

Countries struggle towards meeting commitments to protect biodiversity (Convention on Biodiversity (CBD)) and endangered species (Convention on International Trade in Endangered Species (CITES)), in part because they lack information needed to make informed decisions (Schipper et al. 2008). Little or no information exists on the abundance and distribution of more than 11,000 assessed species (IUCN 2015). This lack of information is especially acute in marine systems (Hamel and Andréfouët 2010, Hansen et al. 2011). To address this information gap, there have been several global efforts to map species distributions, ecosystems, and habitats (e.g. FishBase (www.fishbase.org), Millennium Coral Reef Mapping Project (www.imars.marine.usf.edu/MC/)). Many such maps are created at a coarse or moderate resolution (> 10 m resolution) and thus lack needed detailed habitat information (Andréfouët 2008, IUCN 2012, Roskov et al. 2015). Particularly lacking are high spatial resolution maps of benthic habitats, describing the seafloor's substrates and biotic communities.

The classification and spatial accuracy of maps influences the utility, accuracy, and cost of creation, as well as the representation of features on the map (Wulder et al. 2004, Roelfsema and Phinn 2013). In doing so map accuracy impacts the management decisions made from maps (Gergel et al. 2007, Tulloch et al. 2013). All maps are generalizations of a spatially heterogeneous world and thus inherently contain some misclassification. Acceptable levels of overall map accuracies vary by ecosystem, and level of detail required (e.g. more categories lower accuracies in general), but can be a low as 50 – 60% for coral reefs (Roelfsema and Phinn 2013). Although the overall map accuracy could be high, the individual habitat category accuracy could be lower (e.g. coral classes in general have lower accuracy than bright sand). These differences in accuracy need to be considered when determining the purpose of the map (Roelfsema and Phinn 2013). Although map errors are ubiquitous, they are often overlooked in conservation planning (Langford et al. 2006, Tulloch et al. 2013). Uncertainty in maps comes from many sources including incomplete sampling, measurement errors, processing errors, and a mismatch between the variability of the system and the spatial scale of the map (Gergel et al. 2007, Thompson and Gergel 2008, Roelfsema and Phinn 2008). Ultimately, classification errors impact not only the reported areal extent of any given habitat class, but also impact the perceived arrangement and connectivity among patches (Langford et al. 2006). Variations in perceived arrangement and connectivity of habitats therefore influence any products or decisions that are based on those maps (Gergel et al. 2007, Tulloch et al. 2013).

The effects of map errors on the design of protected area networks are of particular relevance for conserving biodiversity. Protected areas are an important part of conservation strategies because they can reduce the rate of biodiversity loss and can support surrounding, unprotected areas (Margules and Pressey 2000, Almany et al. 2013). As countries work to achieve conservation targets (e.g. CBD Aichi Biodiversity Target to half the loss of all natural habitats by 2020), the spatial extent of marine protected areas (MPAs) has grown at a rate of 4.6% (Wood et al. 2008) with over 1,750 new MPAs created in the past 6 years (Boonzaier 2014). But many existing MPAs were established without maps or based on maps with unknown errors (e.g. Hansen et al.

2011). Ignoring map accuracy in MPA design can lead to omissions of target features, and can reduce the likelihood that MPAs are fully meeting their objectives (Tulloch et al. 2013).

Quantifying the areal extent and spatial arrangement of habitats is important for prioritizing the locations of new MPAs (Grober-Dunsmore et al. 2007, Olds et al. 2012*a*). For example, the availability of suitable habitats influences the distribution and abundance of species (Jennings et al. 1996, Messmer et al. 2011). Where detailed information on species distributions is unavailable, there is empirical evidence that habitat maps can be used as effective surrogates (Margules and Pressey 2000). The arrangement of habitats in the seascape is also significant, particularly where habitat fragmentation is widespread. Habitat fragmentation can affect species by altering the number of suitable patches, increasing the distance between patches, and changing the amount of edge habitat within each patch (Saunders et al. 1991). For example, some species experience higher predation risk near habitat edges (Selgrath et al. 2007) and may benefit from MPAs incorporating locations with less edge habitat. As well, the density of habitat patches can affect the dispersal of organisms (Hovel and Wahle 2010) and influence meta-population persistence (Bengtsson et al. 2003).

Several approaches have been developed to map the spatial composition and extent of marine habitats, including local environmental knowledge (LEK), remote sensing (RS), and in-water habitat surveys. Here I focus on LEK and RS, which are two approaches that have the potential to produce contiguous maps of shallow marine habitats. One aspect of LEK, species and habitat distributions, can be applied to the creation of habitat maps by individuals or focus groups (e.g. Aswani and Lauer 2014). During mapping participants use their expert knowledge to draw maps freely or in combination with satellite or airborne imagery. LEK mapping (also called participatory mapping) has great potential for improving conservation and management by increasing knowledge, complementing scientific measurements, and informing conservation strategies (Thornton and Scheer 2012). Yet the errors and biases in LEK are often unknown, and are important to document (Teixeira et al. 2013). For example, because LEK may focus on practical details (Foale 1998), abundant or visible habitats (Lauer and Aswani 2010), and familiar places (Lauer and Aswani 2008), LEK is rarely evenly distributed across a seascape.

The second mapping approach is remote sensing (RS), which often uses computer algorithms to classify satellite or airborne imagery by assigning map classes to pixels with specific characteristics. RS has the potential to create spatially explicit maps over larger areas with more consistency in coverage than LEK (Lauer and Aswani 2008). However, mapping coral reef and seagrass habitats with RS has long been challenging because of the difficulty of differentiating underwater features that make up the habitats due to water depth and clarity, the presence of different features within one pixel, and the spectral similarity of communities (e.g. coral and algae) (Mumby et al. 1998, Hochberg and Atkinson 2003, Leiper et al. 2012). Technological limitations have largely constrained the approach to identifying geomorphic characteristics (e.g. reef slopes, reef flats) rather than benthic communities, using moderate spatial resolution imagery (pixel sizes 10 m – 100 m; e.g. Millennium Coral Reef Mapping Project) (Andréfouët 2008).

Recent advances in satellites capturing high spatial resolution imagery (pixel sizes 2 m – 10 m) with spectral bands more suitable for marine applications (blue and/or green wavelength ranges, e.g. WorldView-2) have created new possibilities, enabling the creation of benthic community maps of coral reefs at finer spatial scales (2 – 5 m) over large areas (> 300 km^2) (Roelfsema et al. 2013). To date, however, high spatial resolution mapping of coral systems exists in only a handful of areas (Hamel and Andréfouët 2010). One challenge can be the greater cost of high spatial resolution images and the technical expertise required for image processing. These constraints can be particularly limiting for organizations and agencies with limited technical capacity and funding.

Here, I extract and compare a suite of ecological characteristics from maps created using either LEK from coastal fishing communities or RS analysis of high spatial resolution satellite imagery. Our first goal was to understand how each mapping approach depicted the habitat distributions and seascape characteristics of the ecosystem. Our second goal was to explore the conservation implications and costs that influence which mapping approach is most appropriate for different situations.

4.2 Methods

4.2.1 Benthic community map creation

4.2.1.1 Overview

I created and compared benthic habitat maps of the central Danajon Bank (Figure 4.1) based on two mapping approaches: LEK and RS mapping. Both approaches involved field data collection, pre-processing of field and/or satellite image data, determination of appropriate benthic community class divisions, and map creation. The LEK approach used participatory

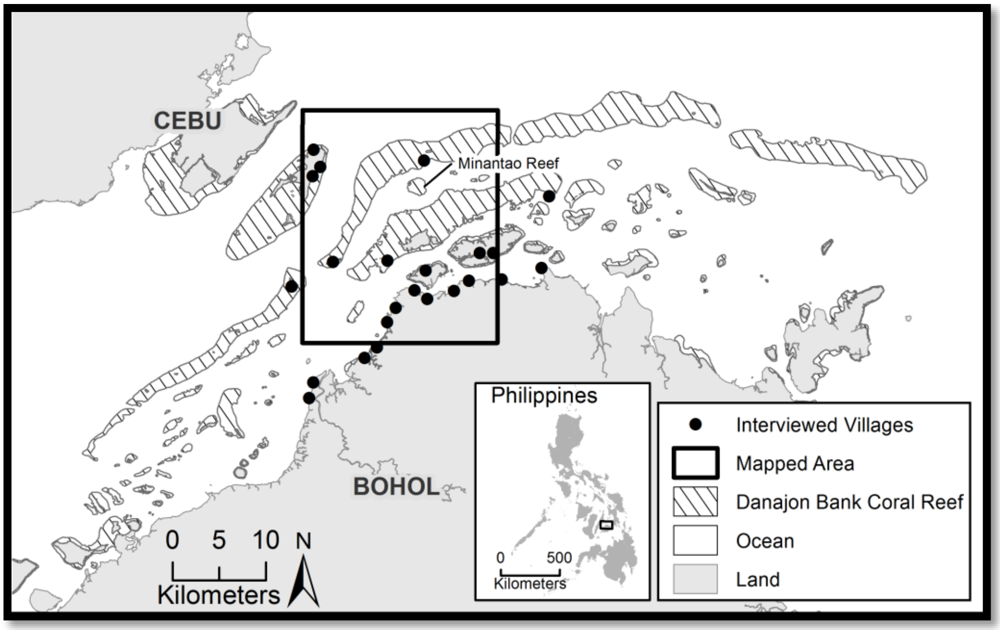

Figure 4.1 The study area for which local environmental knowledge (LEK) and high spatial resolution remote sensing (RS) maps were created is located in the central Danajon Bank, Philippines. Villages where interviews took place were stratified by location in the ecosystem. Validation surveys took place at the Minantao Reef.

mapping to delineate habitats incorporating SPOT-5 satellite image as a basemap, while the RS approach classified two WorldView-2 satellite images using the full spectral characteristics of the different bands that make up the imagery within an object-based classification. Mapped classes were based on a composite of the ecological relevance of habitats as well as technological constraints in distinguishing among complex coral reef habitats, which often occur as heterogeneous, highly-mixed mosaics (Capolsini et al. 2003). For RS and LEK mapping approaches I define benthic habitats to include abiotic substrates (e.g. sand) and biotic communities (e.g. coral) growing on the seafloor. Habitat classes used here comprised a mix of five benthic cover types (coral, rubble, sand, seagrass, and algae). Although germane to reef health and conservation, I excluded mangroves from comparisons because the LEK mapping method used here focused solely on fishing grounds, rarely located in mangrove habitats (See LEK Data Collection). Deep sea water, clouds, and land were also excluded.

4.2.1.2 Local environmental knowledge mapping approach

4.2.1.2.1 LEK field data collection.

LEK comprises the integrated and situated knowledge, practices, and beliefs of communities and resource users regarding the local environment and their relationship with it (Berkes 2012, McMillen et al. 2014). To map LEK of benthic habitats I conducted participatory mapping interviews in the local language, Cebuano. I interviewed 249 fishers from 20 villages between July 2010 and April 2011. See Appendix A for interview questions. Villages were randomly selected and were stratified by their location (e.g. mainland, large islands, cays) to include geographically contrasting parts of the reef system. I asked fishers to identify their current fishing grounds in a 20 by 25 km area and to describe the habitats therein (Figure 4.2). To make the maps as spatially precise as possible, I drew habitat boundaries over a georeferenced SPOT-5 satellite image (4 bands: green, red, NIR, MIR; 10 m x 10 m pixel size), which was the highest resolution image available at the time interviews were conducted. The high spatial resolution WorldView-2 images used for the RS mapping (see below) were not available at the time of the LEK fieldwork. This integration of technology (similar to aerial photography interpretation) allowed fishers to orient their drawings to geographic features in the seascape and to incorporate the texture and color of the image into their mapping (Morgan et al. 2010). Since many respondents were unfamiliar with satellite images and maps, I oriented fishers to the map and confirmed their ability to identify locations and features on the map before collecting any data. I offered respondents a list of nine habitat classes with photos to help standardize responses, although some fishers provided four other habitat categories (e.g. *bato* (Cebuano for rocks, but a term used to classify both coral and rubble); *taganas* (Cebuano for deep areas that are adjacent to the reef slope)).

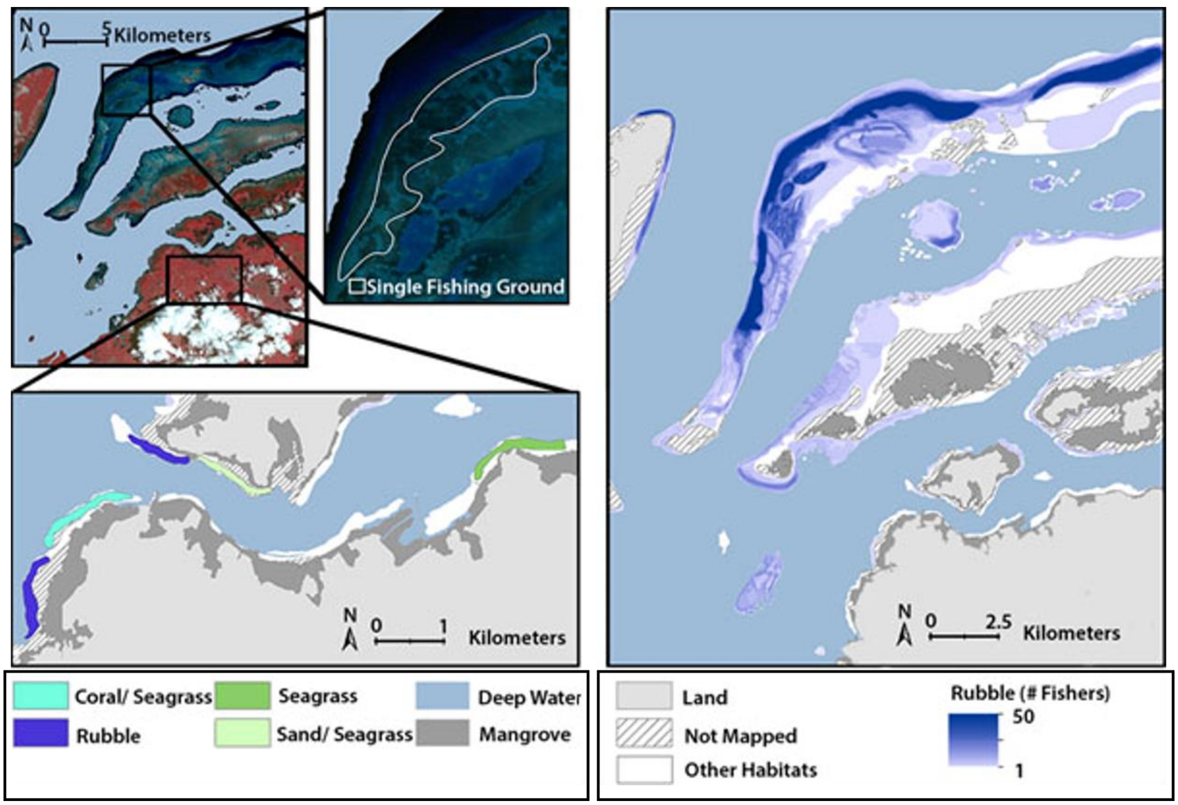

Figure 4.2 The process of creating LEK maps. (a) Fishers mapped habitats on a SPOT-5 satellite image base map (10 m x 10 m pixel size); (b) Fishers delineated single fishing grounds and mapped the habitats found therein; (c) Each fisher created a habitat map which identified the habitats found in all of their fishing grounds; (d) For each benthic habitat that was mapped, I layered the maps for all fishers and counted the number of fishers who documented that a habitat was present at each location on the map. Depicted here is the map showing how many fishers said 'Rubble' was present.

4.2.1.2.2 LEK based classification.

I created a map representing fishers' cumulative LEK by layering habitat maps from all respondents into one map showing the most commonly identified habitat (Figure 4.2). To achieve this, I first digitized the maps drawn during each interview using heads-up digitizing in ArcGIS 10.1 (Environmental Systems Research Institute, Redlands, California, USA). I then calculated the number of respondents that identified a given habitat at each location on the map (i.e. in each grid cell; Figure 4.2). The final habitat class included the dominant and sub-dominant habitats which were reported by the highest and second highest number of respondents, respectively (e.g. if seven fishers reported 'Coral' and six fishers reported 'Rubble' the habitat class for that location would be 'Coral/Rubble' (Table G.1). Calculations used ArcGIS 10.1 and R 2.15.2 (package: Raster) (Hijmans and Etten 2014, R Core Team 2014). I modified existing R commands in the Raster package to calculate the habitats reported by the highest and second highest number of fishers. R scripts for conducting this analysis are included in the supplement.

I made four alterations to the LEK data: (i) I simplified LEK habitat classes by combining rare habitat classes (< 200 ha) with ecologically similar classes; (ii) I merged all polygons smaller than 100 m2 with neighboring polygons; (iii) I filled small gaps in coverage (< 1.5 ha) with neighboring polygons; and (iv) I assigned larger gaps to a 'No Data' category. Gaps in the LEK maps occurred because fishers mapped their fishing grounds, but not the surrounding areas.

4.2.1.3 Remote sensing mapping approach

4.2.1.3.1 RS field calibration data.

To create the RS map, I conducted benthic cover (seafloor) field surveys using two methods. First, I undertook georeferenced point intercept transects (English et al. 1997). For 11 sites I recorded habitat cover types at 0.5 m intervals on 20 m long transects (n= 2,070 points with 6 transects at most sites (range 3 - 8). I distributed the transects across as many habitats as possible at each site, allowing us to obtain a representative sample of the habitats. Second, I conducted geo-referenced spot-check surveys, by placing a viewing bucket in the water to estimate the percent cover benthic cover types (n = 2,357 points). Survey locations were chosen to cover a diverse and representative subset of habitats found in the Danajon Bank. The combination of methods was a compromise between the higher accuracy of point intercept transects and the larger sample area achievable through spot-check surveys (Roelfsema and Phinn 2008).

4.2.1.3.2 RS image acquisition and pre-processing.

To cover the full extent of the study area, I acquired two WorldView-2 images (05/10/2010 and 20/04/2012) from the Digital Globe archive. These images were selected for having the lowest cloud cover and the shortest time lag between them. The WorldView-2 sensor has 8 multi-spectral bands (coastal, blue, green, yellow, red, NIR, MIR1 and MIR 2) with a 2 m x 2 m pixel size. I initially created a RS map using a pixel-based classification of the same SPOT-5 satellite image as the LEK mapping, but after obtaining map accuracies < 50% I switched to the WorldView-2 images used here. The WorldView-2 images were radiometrically and geometrically corrected by DigitalGlobe with a stated accuracy of 5 m (www.digitalglobe.com). The two images were dark pixel corrected (Jensen 2005) and joined to form an almost seamless mosaic. I used WorldView-2 imagery because past studies have shown that they are suitable to create habitats maps using object-based image analysis (Phinn et al. 2012, Roelfsema et al. 2013b). All image pre-processing was conducted using ENVI 5.0 (Exelis Visual Information Solutions, Boulder, Colorado) and ArcGIS 10.1 software.

4.2.1.3.3 RS based classification.

I used object-based image analysis (Blaschke 2010) for the RS classification of the WorldView-2 mosaic following the same approach as explained in detail in Phinn et al. (2012) and Roelfsema et al. (2013). This technique classifies the image into maps with increasing detail, resulting in a hierarchical classification. Where pixel based classification assigns a class to each individual pixel, object-based image analysis segments the image in groups of pixel with same colour and texture and then assigns a label to each segment following predefined decision rules. Decision rules for classifying images are included in the supplement. These decision rules are based on the segments' colour, texture, and contextual relationships. Contextual relationships can include: other hierarchical levels (e.g. geomorphology can influence benthic habitat classes) or spatial proximities to other classes (Mumby et al. 1998). The RS method I used is well suited to classifying high spatial resolution imagery, because pixel variance is grouped into image-objects approximating real features (Blaschke 2010).

I classified the image mosaic using three hierarchical levels of image-objects: reef, geomorphic, and benthic community using a 4 m^2 minimum mapping unit. The reef level distinguished reef, land, deep water, and clouds; while the geomorphic level classified reef slope, inner reef flat, outer reef flat, mangroves, re-planted mangroves, deep water, mainland, terrestrial islands, cays, and clouds (Table G.2) (Roelfsema et al. 2013b). The benthic community level segregated the final image into 17 classes. Object-based image analysis was conducted using eCognition 8.4 (Trimble, Sunnyvale, California, USA). To make the RS and LEK approaches directly comparable, I clipped the RS map to the smaller area of the LEK map (Table G.2).

4.2.2 Comparisons of LEK and RS based approaches

I evaluated the results of both mapping approaches in five ways: (i) a quantitative map validation based on independent habitat surveys; (ii) a quantitative assessment of the agreement between the maps; (iii) a qualitative comparison of the maps; (iv) a quantitative assessment of how each map characterized seascape characteristics; and (v) a comparison of mapping costs.

4.2.2.1 Quantitative map validation

4.2.2.1.1 Field validation data

Independent manta tow survey data (English et al. 1997) was used to assess the accuracy of the maps for a subset of the study area. These surveys took place at a 4 km^2 reef (Minantao) located near the center of the map (Figure 4.3). Although a validation survey that sampled many parts of the study area would have been preferred, due to limited resources the validation survey was only available for this subset of the mapped area. The independent validation data was considered representative for the whole study area. The manta tow survey documented four major habitats: ‘Coral’, ‘Seagrass’, ‘Rubble’, and ‘Sand’, the same as used for the major habitat classes in the LEK or RS based maps. ‘Patchy Coral,’ also documented by the surveys, was not included in the accuracy assessment because I was not able to assign these manta tow points to a dominant habitat class in the maps. I also excluded manta tow points located in ‘Deep Water’ on the maps. Because more manta tow points fell into ‘Deep Water’ areas on the LEK map, there were 19 fewer manta tow points for evaluating the LEK map.

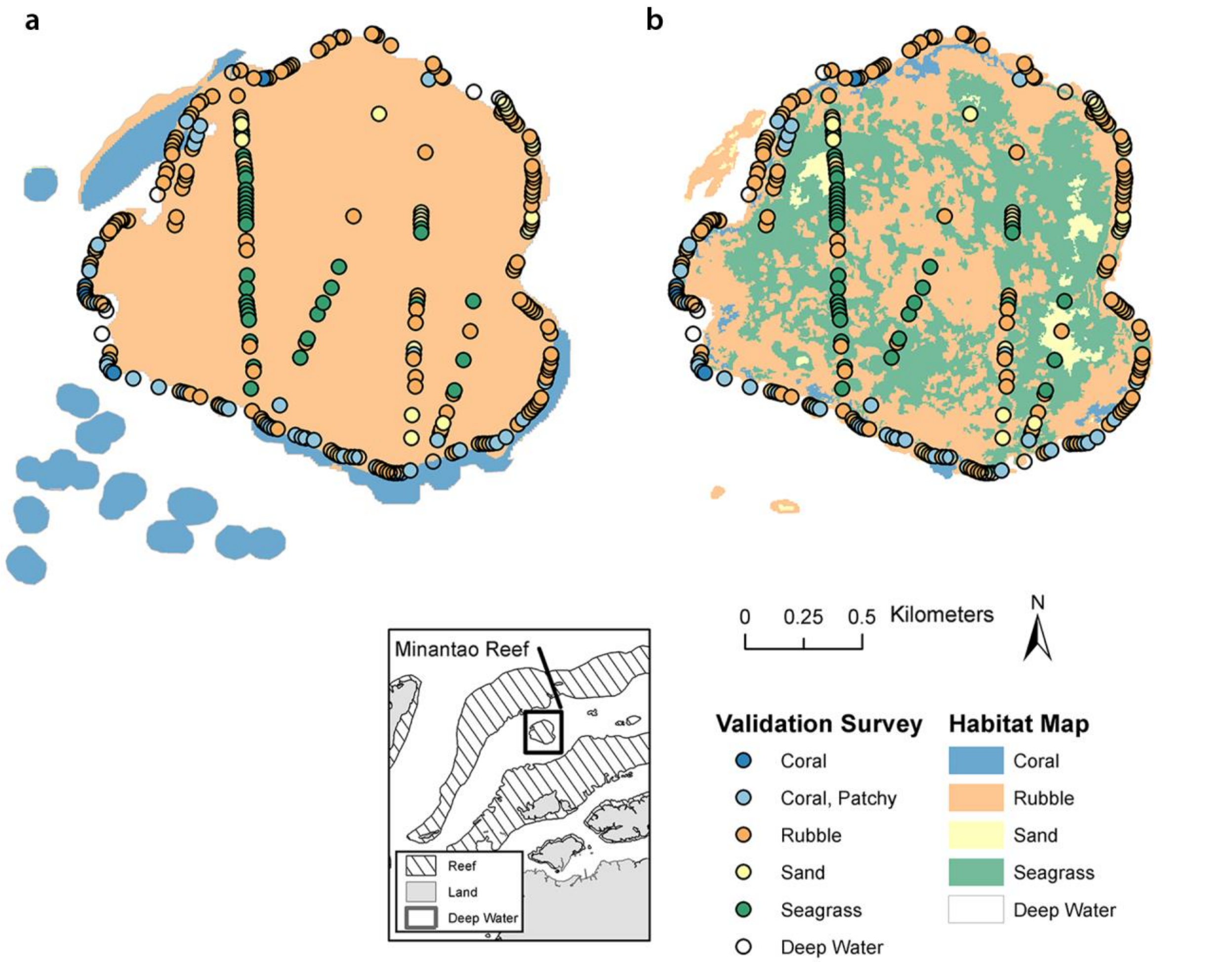

Figure 4.3 Validation for a subset of the Danajon Bank habitat maps used independent habitat survey data from the Minantao Reef. Validation surveys are shown overlaying the: (a) local environmental knowledge (LEK) map; and (b) high spatial resolution remote sensing (RS) map.

4.2.2.1.2 Accuracy assessment

For each map I compared the field validation to the mapped habitat classes. An error matrix was created from the reference and mapped data, to calculate the overall map and individual class accuracies (Congalton 1991). Overall accuracy (the percentage of points that were classified correctly) estimates the overall reliability of the classification. Producer and user accuracy were calculated for the individual map classes. Producer's accuracy (error of commission) is the probability that a point on a map is correctly categorized by the classification scheme; while the user's accuracy (error of omission) estimates the probability that the class assigned to a point on the map accurately represents what is on the ground. Overall map and individual map class accuracies can be influenced by the size of the area mapped, the habitat complexity, the number of habitat classes, and the mapping approach (Roelfsema and Phinn 2013). As approximately 60% accuracy is the standard for marine remote sensing maps (Roelfsema and Phinn 2013), I considered anything higher than 60% agreement to indicate a good fit.

The reference data were opportunistically collected for another project using the manta tow technique (Panes and Nellas 1997), hence not all habitat classes were surveyed that were also present in the LEK or RS habitat maps (e.g. a 'Sand/Seagrass' class was mapped using LEK and RS approaches, but was not included in the Manta Tow survey). Thus I simplified the mapped habitat classes to match the validation survey classes, which focused on major habitat classes. This led us to have two sets of maps: the original maps and map incorporating the major habitat classes (i.e. corresponding to the validation survey habitat classes). Unless otherwise stated, analyses were conducted on the major habitat class map.

4.2.3 Quantitative map agreement

To assess agreement between the LEK and RS maps, I sampled the mapping categories for both maps at 1,000 randomly sampled points. Points were restricted to shallow areas where benthic habitats were categorized by both mapping approaches. I used the sampled values to create a summary matrix, which included the overall agreement and the per-category agreements. I took this approach because I assumed neither map represented 'the truth' (Morgan and Gergel 2013). The quantitative map comparison was conducted twice: once with the major habitat classes used in the manta tow survey; and once using the original habitat classes. For the later comparison, I considered 'matches' to include agreement between any of the habitats mapped at a location (e.g. 'Seagrass' was considered to match 'Sand/Seagrass'). I found that this method was the most concordant way to address the fact that the habitat classes in the LEK map and the RS map were not identical.

4.2.4 Qualitative visual assessment

To further understand the maps, I documented qualitative differences between the mapping approaches, including geographic variations in benthic habitat distributions, and the local areas where habitats were not successfully mapped. This was especially important to assess areas that were not part of the validation.

4.2.5 Seascape characteristics assessment

I evaluated how the LEK and RS mapping approaches quantified three seascape characteristics: habitat abundance; heterogeneity; and connectivity. The two mapping approaches covered the same 20 km x 25 km area. However, the final versions of the RS and LEK maps classified benthic habitats in a slightly different area (e.g. the LEK map had gaps where no respondents mapped habitats; see Results). To account for this difference, I used metrics of seascape characteristics that were standardized by the area (ha) where habitats were classified.

Habitat abundance was quantified by evaluating the percent of the seascape (aka landscape) covered by each habitat class. Here, I defined ‘seascape’ to include the area (ha) of a map where benthic habitats were classified, and to exclude deep water, land, mangroves, and unclassified areas. Heterogeneity was measured using two landscape indices: Patch Area and Edge Density. Patch Area measures the mean size of habitat patches in the seascape, while Edge Density measures the linear length of edge (in meters) per ha. I defined an edge as the border between two adjacent habitat classes and did not distinguish natural and anthropogenic habitat edges. I chose Edge Density because habitat edges can influence species distributions and survival rates (Selgrath et al. 2007).

To quantify the seascape’s structural connectivity (the physical attributes of the seascape, which theoretically influence the ability of species to disperse; hereafter ‘connectivity’) (Calabrese and Fagan 2004, Grober-Dunsmore et al. 2008), I used Patch Density (number of habitat patches per 100 ha) and Near-Neighbor Distance (the shortest distance between two patches of the same habitat class). Theoretically, connectivity increases with Patch Density, due to the greater ability of individuals to disperse through a seascape. The actual connectivity depends on the species of interest. A smaller Near-Neighbor distance suggests higher connectivity in the seascape. Landscape metrics were calculated in Fragstats3 (McGarigal et al. 2012) and R (2.15.2).

4.2.6 Cost estimation

Since cost can be a substantial determinant of mapping feasibility, I compared the costs of the two mapping approaches. Fieldwork for both approaches was a subset of other projects. Therefore I estimated the number of hours that would be needed for directly working on the mapping for those who were already familiar with a site and with existing expertise in fieldwork and object-based image analysis. I assumed a small NGO or government agency in a developing country conducted field surveys. For this research project, the satellite imagery was donated, and I obtained a discounted license for the object-based image analysis software (eCognition). Thus, for the cost estimate, I used the standard costs of these items if obtained without discounts, and I assumed NGOs would use qGIS (a free GIS software; QGIS Development Team, Open Source Geospatial Foundation Project).

4.3 Results

4.3.1 Summary of LEK and RS based map creation

I created 20 km x 25 km habitat maps of the study area (Figure 4) and I found that both mapping approaches were able to characterize the coral reef habitats accurately. This assessment was based on the quantitative map assessment for the Minantao Reef (Figure 1; Figure 4.3), for which the validation resulted in acceptable level of accuracy (> 60%; Roelfsema and Phinn 2013). Overall the RS map performed better than the LEK map, although each mapping approach had various strengths and weaknesses, detailed below and summarized in Table 4.1.

Within the 20 km x 25 km study area, both maps included habitats that were not evaluated (e.g. land) and had some locations that were left unclassified. The RS map had gaps due to cloud cover. The LEK map had gaps at locations where no respondents identified habitats. As a result, the total amount of classified shallow habitat differed slightly for each map. The LEK map classified 22% (10,902 ha) of the 50,000 ha study area as benthic habitats, while the RS map classified 28% (13,865 ha).

On average LEK habitat patches were five times as large as RS habitat patches (Table 4.1), indicating that the RS approach provided higher spatial precision for habitat locations. The LEK approach produced maps with 18 habitat classes (Table G.1; see Methods for details). The RS approach produced maps with 16 benthic habitats (Table G.2; see Methods for details).

4.3.2 Comparisons of LEK and RS based approaches

4.3.2.1 Quantitative map validation

For the accuracy assessments of each mapping approach, the RS map outperformed the LEK map (LEK = 66%; RS overall accuracy = 76%; Table 4.2). Both mapping approaches exceeded the minimum mapping standard of 60% (Roelfsema and Phinn 2013). The LEK map correctly identified all 'Rubble' (producer's accuracy = 100%), but failed to capture other habitat classes. The high LEK accuracy of 'Rubble' suggests that this class was over-mapped with the LEK approach. In the area where validation surveys took place, 'Sand' and 'Seagrass' were not included in the LEK maps, although they were mapped elsewhere. When I assessed the accuracy of the RS map, I found that 'Rubble' and 'Seagrass' were mapped more consistently than 'Coral' and 'Sand.' For both LEK and RS approaches, it was difficult to assess 'Coral' as only 1 'Coral' reference point overlapped the LEK map and only 5 'Coral' reference points overlapped the RS map (Table 4.2; Figure 4.3).

Table 4.1 Summary table of findings for comparison of local environmental knowledge (LEK) and high resolution remote sensing (RS) mapping approaches.

	Mapping Approach		
Metric	**LEK**	**RS**	**Comment**
Overall map accuracy	66%	76%	Accuracy higher than the required 60% for both approaches.
Class accuracies	Over classifying 'Rubble'; under classifying 'Sand' and 'Seagrass'.	Higher accuracies then LEK for 'Rubble' and 'Seagrass'.	The remote sensing map had higher class accuracies and did a more consistent job of mapping habitats.
Quantitative map agreement			Maps have a 37% agreement for major habitat classes. Agreement varies between habitat classes. Map agreement is higher (62%) when evaluating original habitat classes.
Qualitative map agreement	Missed habitats where no respondents fished. Better at classifying habitats in turbid waters.	Mapped finer detail of habitat arrangements.	Each approach has different strengths.
Habitat distribution	'Rubble' and 'Coral' dominated.	'Seagrass' dominated.	Different habitats dominated each map.
Landscape indices			
Patch Area (ha, mean ± SE)	8.8 ± 2.5	1.7 ± 0.4	The remote sensing approach identifies smaller habitat patches.
Edge Density (length of edges per m^2)	9.5	51.4	Remote sensing provides more detail about habitat edges.
Connectivity			
Near Neighbor Distance (m; area mean ± SE)	37.6 ± 5.1	10.3 ± 0.9	Remote sensing maps depict 3.7 to 6.4 higher connectivity due to a shorter distance between habitat patches and a higher density of habitat patches.
Patch Density (no. Patches per 100 ha)	2.4	15.4	
Cost			
Donated images and software	$7,718	$10,188	Remote sensing maps are more expensive and require greater technical skill, but some costs can be offset.
Purchased images and software	$9,343	$47,688	

4.3.2.2 Quantitative map agreement between LEK and RS

The comparative agreement between the two mapping approaches showed that maps had 37% overall agreement when using the major habitat classes (Table G.3a; Figure 4.4). When I instead compared the maps using the original classes (Figure 4.5), the map agreement was higher (62%; Table G.3b). Among the two mapping approaches there was wide range in agreement between various classes. For example, 'Seagrass' was mapped inconsistently between the two maps.

4.3.2.3 Qualitative visual assessment

A visual overview of the two mapping approaches reveals that the maps are fairly different. Their differences were most pronounced in the outer reef where the LEK primarily mapped 'Rubble' while the RS primarily mapped 'Seagrass' (Figure 4.4; Figure 4.6). When considering qualitative differences between the maps, there were various strengths of each mapping method (Figure 4.4; Table 4.1). The RS map was able to pick up much finer-scale detail in the study area. In locations where respondents did not fish, the LEK map had no data while the RS map was able to identify habitats. In inner areas with relatively turbid waters, the RS map missed seagrass beds that were mapped using the LEK method.

4.3.2.4 Seascape characteristics assessment

When evaluating how the two mapping approaches characterized major habitat distributions, both the maps showed that habitats covered a different proportion of the seascape and were distributed in different ways (Table 4.1, Figure 4.5). Measures of habitat heterogeneity showed that 'Rubble' and 'Coral' dominated the LEK map, covering 44% and 34% of the LEK seascape,

Table 4.2 Confusion matrices comparing independent habitat survey data from manta tows with the (a) local environmental knowledge (LEK) and (b) remote sensing (RS) maps at the Minantao Reef.

(a)

LEK Map	Reference Habitats				Total
	Coral	**Rubble**	**Sand**	**Seagrass**	
Habitats					
Coral	**0**	0	1	0	1
Rubble	1	**121**	23	37	182
Sand	0	0	**0**	0	0
Seagrass	0	0	0	**0**	0
Total	1	121	24	37	183
Accuracy					
Producer's Accuracy	0%	100%	0%	0%	
User's Accuracy	0%	66%	0%	0%	
Overall Accuracy					66%

(b)

RS Map	Reference Habitats				Total
	Coral	**Rubble**	**Sand**	**Seagrass**	
Habitats					
Coral	**1**	2	0	0	3
Rubble	3	**117**	12	3	135
Sand	1	2	**2**	1	6
Seagrass		12	13	**33**	58
Total	5	133	27	37	202
Accuracy					
Producer's Accuracy	20%	88%	7%	89%	
User's Accuracy	33%	87%	33%	57%	
Overall Accuracy					76%

Note: I excluded reference (manta tow) points located in 'Deep Water' on the maps. Since more reference points fell into 'Deep Water' areas on the LEK map, there were 19 fewer manta tow points for evaluating the LEK map (LEK n = 183; RS n = 202).

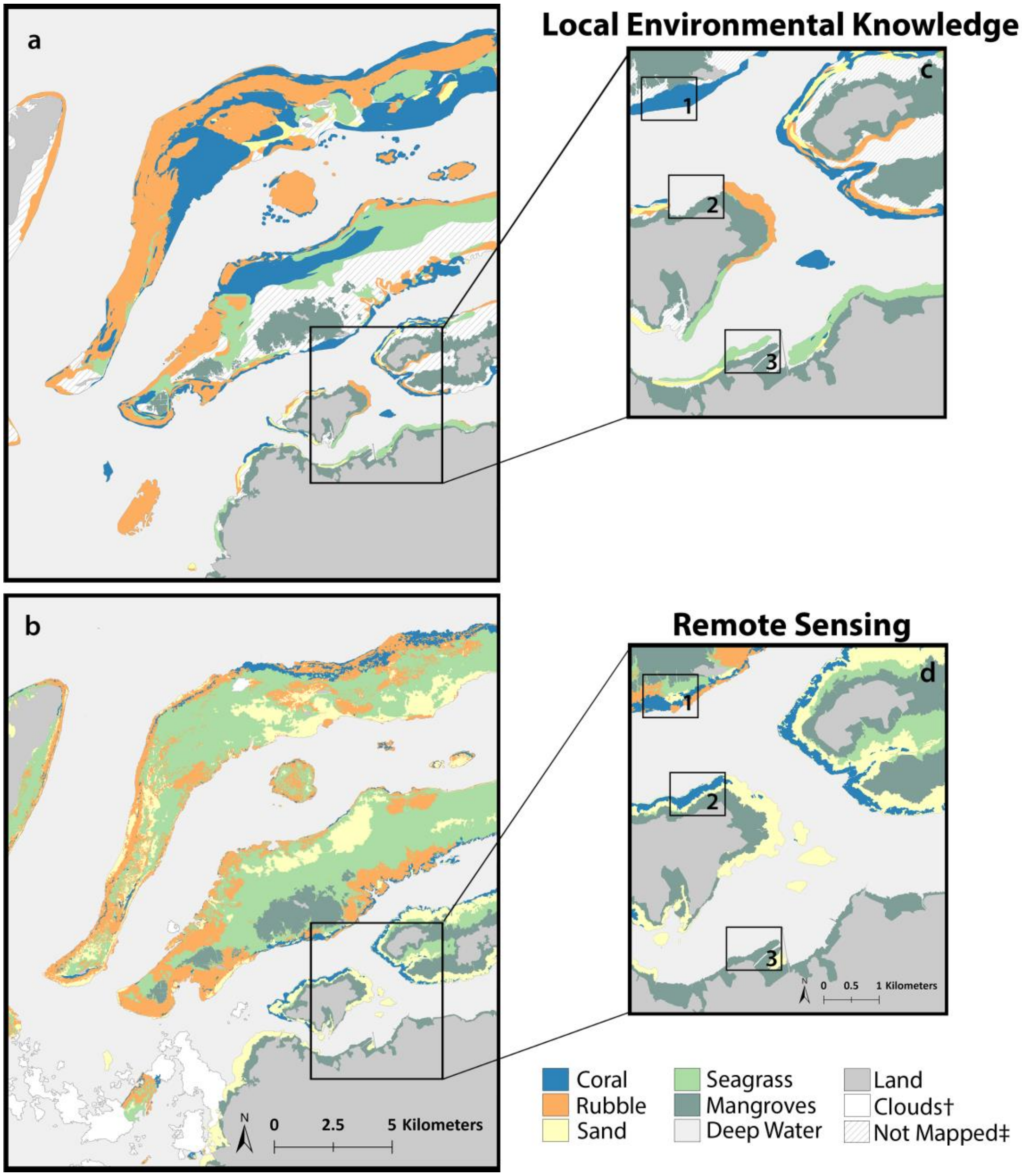

Figure 4.4 Maps of major habitat classes for: (a) local environmental knowledge (LEK) and (b) high spatial resolution remote sensing (RS) habitat maps of benthic habitats in the Danajon Bank, Philippines. Close up examples demonstrate differences in benthic habitat maps made using (c) LEK and (d) RS. In some areas: the LEK approach shows less detail than the RS map (c1 & d1); the LEK approach missed habitats that the RS approach captured (c2 & d2); or the LEK approach mapped habitats that the RS maps missed (c3 & d3). †= classes unique to LEK map; ‡= classes unique to RS map.

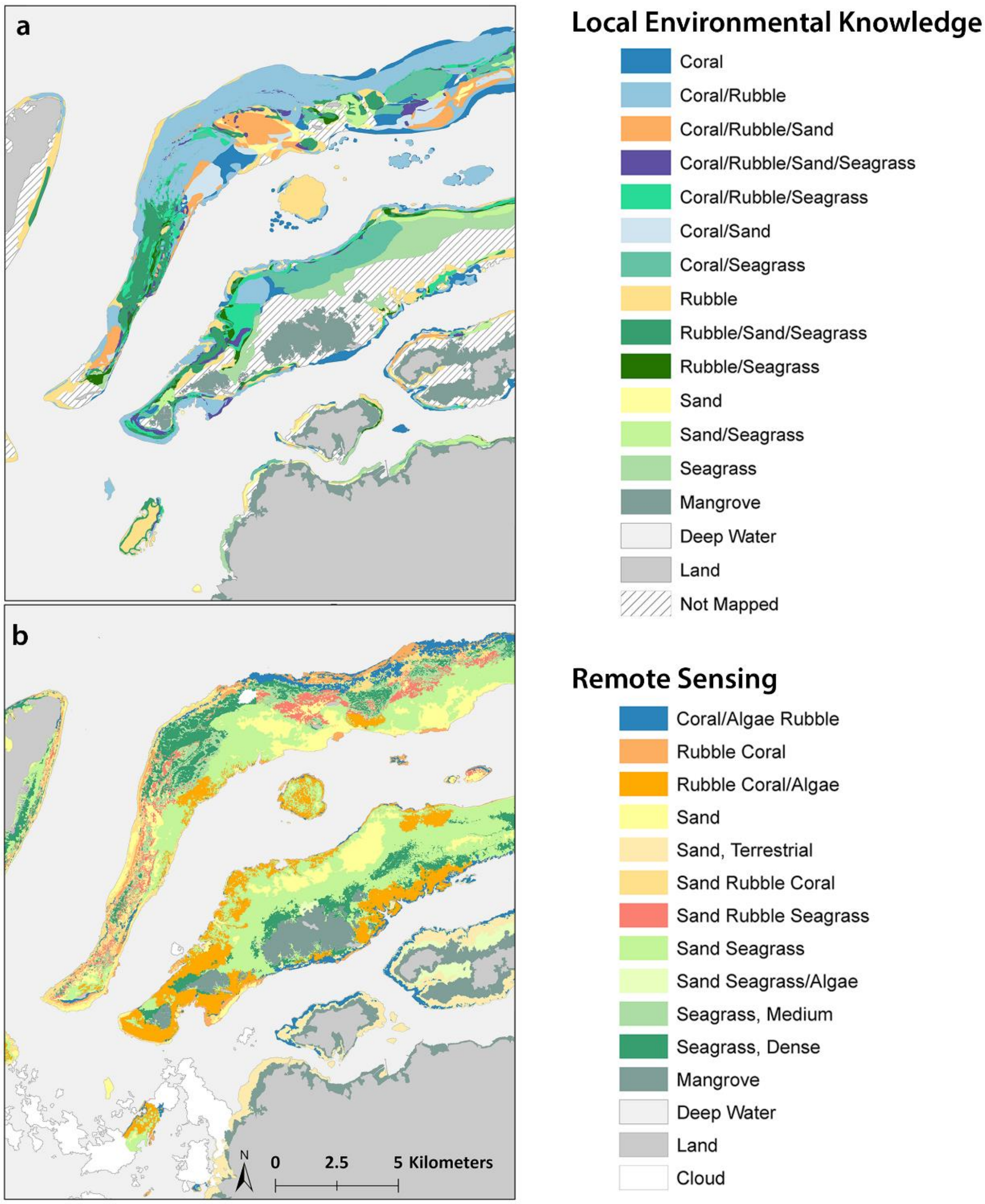

Figure 4.5 Detailed habitat classes for: (a) local environmental knowledge (LEK) and (b) high spatial resolution remote sensing (RS) habitat maps of benthic habitats in the Danajon Bank, Philippines. I developed LEK maps by participatory mapping with small-scale fishers and RS maps by using object-based image analysis to classify WorldView-2 satellite images.

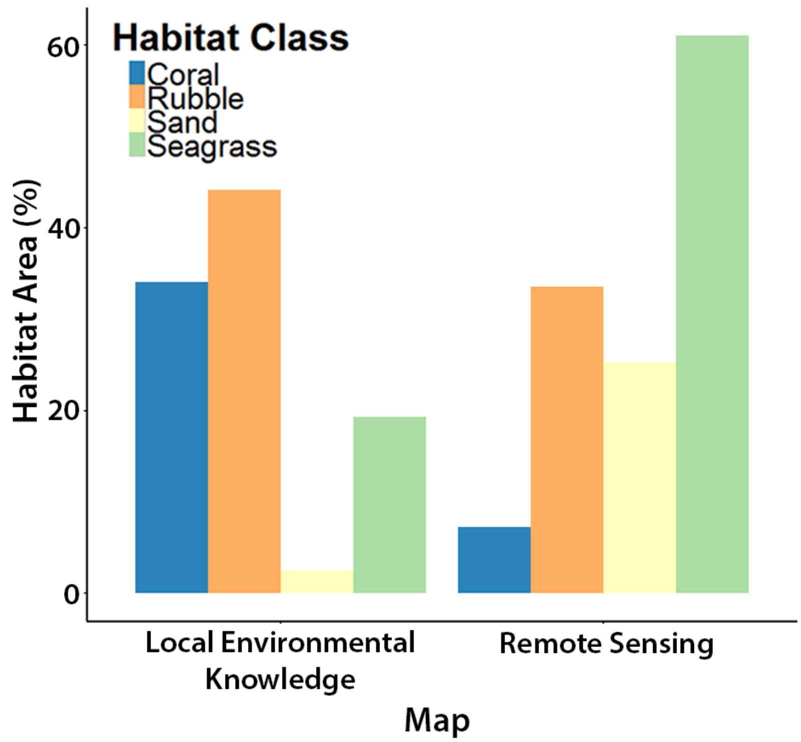

Figure 4.6 Contrasting estimates of percent habitat cover of the Danajon Bank, Philippines based on habitat maps created using local environmental knowledge (LEK) and high resolution remote sensing of WorldView2 satellite images. These estimates are based on major habitat classes, which match the validation survey classes.

respectively. In contrast 'Seagrass' dominated the RS map, covering 48% of the RS seascape. The RS approach found low 'Coral' cover (6%) while the LEK approach showed higher 'Coral' cover (34%). In the original RS map (i.e. the map that was not simplified to match the validation survey categories), there was a lower percent cover of categories containing coral (Figure 4.7). The two mapping approaches characterized the seascape as having quite different habitat heterogeneity and connectivity (Figure 4.8). The RS map depicted greater complexity in the spatial arrangement of benthic habitats, as characterized by having six times more habitat patches (Table 4.1). The RS approach distinguished approximately four times as many edges between adjacent habitats (Table 4.1). The finer resolution of the RS map was thus better at documenting the full extent of habitat edges. Habitat connectivity was five times higher with the RS approach than in the LEK map when measured using Patch Density (Table 4.1) and four times higher in the RS map when measured using Near Neighbor Distance (Table 4.1). Note that a higher Near Neighbor Distance indicates a higher distance between similar patches and lower connectivity. Thus the RS approach suggested higher structural connectivity in the seascape and indicated that species (e.g. fish, crustaceans) traveling across the seascape could travel shorter distances between patches of the same type of habitat (e.g. between two 'Coral' patches).

4.3.2.5 Cost estimation

Based on our cost estimation, the RS map was approximately one and a half times as expensive to produce for this project when images were donated and software was available from existing licenses (total cost: $7,718 LEK and $10,188 RS; Table 4.3). When imagery and software would need to be purchased, I estimate that the RS map would be five times more expensive then the LEK map (total cost: $9,343 LEK and $47,688 RS; Table 4.3). Ideally the software could be used for several projects. The primary cost for the LEK map was time (83%), while the primary cost of the RS method was the high spatial resolution satellite images and software (79%).

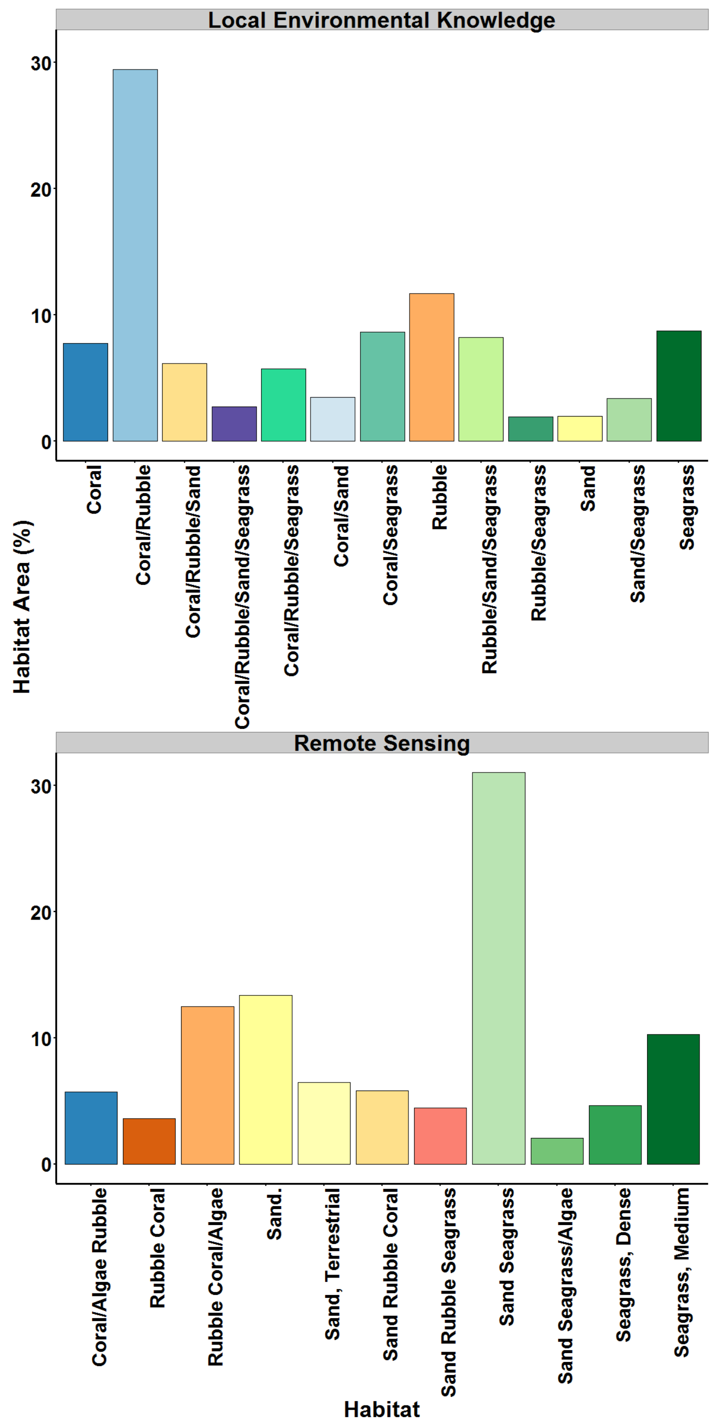

Figure 4.7 Contrasting estimates of percent habitat cover of the Danajon Bank, Philippines based on habitat maps created using (a) local environmental knowledge and (b) high resolution remote sensing of WorldView2 satellite images. These estimates are based on the habitat classes from the detailed maps. See text for details.

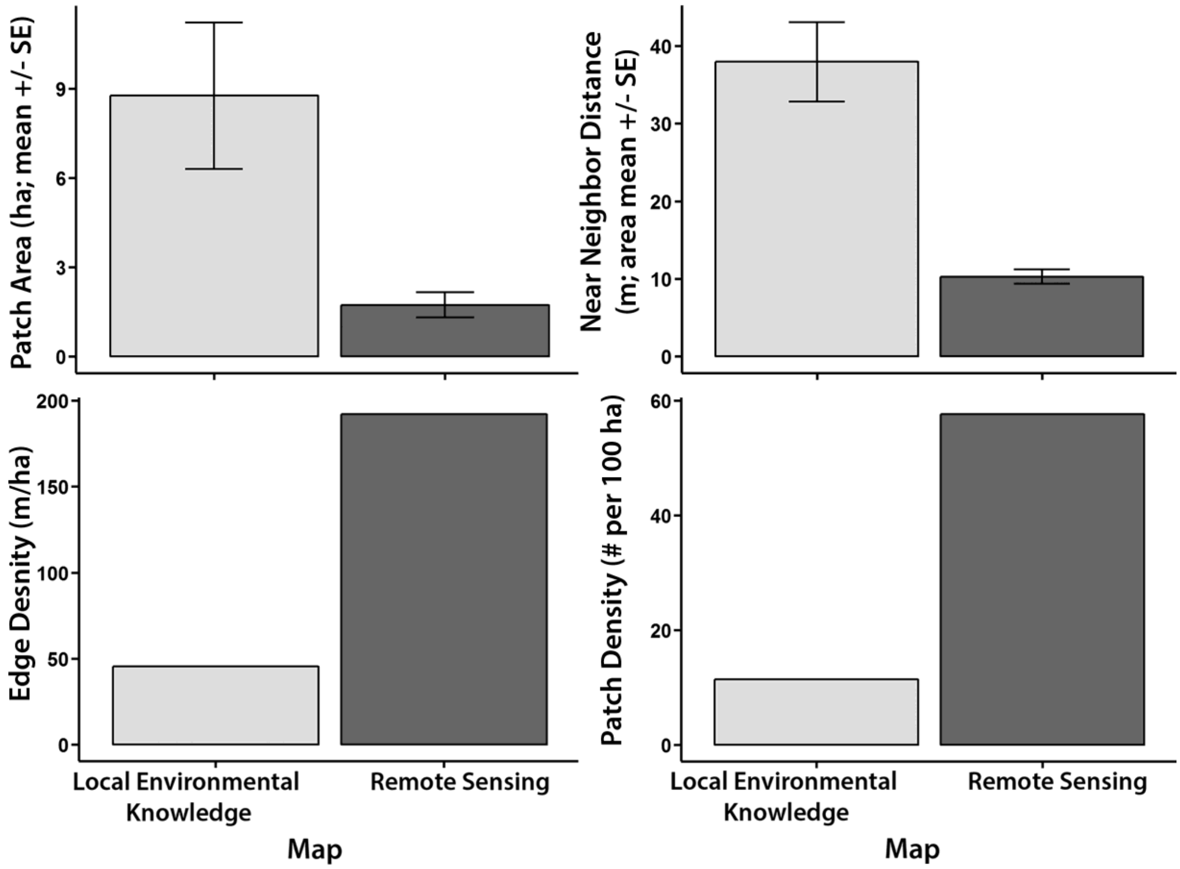

Figure 4.8 Contrasting estimates of seascape characteristics of the Danajon Bank, Philippines bases on maps created using local environmental knowledge (LEK) and classification of high spatial resolution remote sensing (RS) of WorldView-2 images. The seascape estimates used here incorporate information from all habitat classes. (a) Area of habitat patches; (b) Distance between a habitat patch and the nearest patch of the same habitat class; (c) Density of habitat edges (the borders between two different types of habitat); (d) Density of habitat patches.

Table 4.3 Estimated costs (USD) using local environmental knowledge (LEK) and remote sensing (RS) approaches for mapping benthic community habitats.

Task	Cost Per Hour	LEK			RS		
		Person-hours	Other Costs	Total Cost	Person-hours	Other Costs	Total Cost
Establishing contacts, identifying field sites	6	40	$100	$340	20	$100	$220
Field surveys (interviews, habitat surveys)	6	315	$400	$2,290	160	$700	$1,660
Validation surveys	6	48	$300	$588	48	$300	$588
Data entry (including digitizing)	6	100		$600	40		$240
Obtaining advice from previous studies (r code, classification rules; technician)	20	40		$800	24		$480
Data processing & map creation (technician)	20	80		$1,600	200		$4,000
Supervision, data processing & map creation (research associate/ professor)	150	10		$1,500	20		$3,000
Total cost, without images or software		*633*	*$800*	*$7,718*	*512*	*$1,100*	*$10,188*
Satellite images			$1,625	$1,625†		$17,500	$17,500 †
Software		$0	$0	$0‡		$20,000	$20,000 §
Total cost		*1,266*	*$2,425*	*$9,343*	*512*	*$38,600*	*$47,688*

Note: The LEK costs assume an existing NGO has established community contacts and staff skilled with ecological and LEK fieldwork. The RS costs assume that the technician has technical expertise in object-based image analysis.
† Images for this project were donated through Planet Action grants, but these are the estimated image costs for SPOT-5 and WorldView-2.
‡ If free software, qGIS and R, are used for the project.
§ If eCognition is used and an independent license is purchased.

4.4 Discussion

Large coral reef systems have rarely been mapped using high-spatial resolution satellite imagery (Roelfsema et al. 2013b). Thus this study, by mapping a 500 km^2 area, provides a rare quantitative assessment of the accuracy and seascape characteristics of RS and LEK mapping at this scale. The accuracy of the RS map, similar to the accuracy of other studies (Roelfsema et al. 2013), provides evidence that this object-based mapping approach can be successfully applied in a new region (South-East Asia). The RS mapping approach used here was originally developed for the Western Pacific (Phinn et al. 2012, Lyons et al. 2012, Roelfsema et al. 2013b). Furthermore the current study demonstrates that although the remote sensing specialist is not familiar with the reefs in the study area, accurate and reliable RS maps can be created through the presence of: expert knowledge; sufficient field data; high spatial resolution imagery; and existing rulesets developed for other reefs (Roelfsema et al. 2013b).

Our detailed evaluation of LEK and RS approaches enabled us to test the often implicit assumption that RS maps are more accurate than LEK maps. I show that the RS approach was indeed more accurate, but both methods met the 60% standard for overall accuracy (Roelfsema and Phinn 2013). Our LEK map was in a similar overall accuracy range as coral reef habitat maps produced by other studies, including moderate spatial resolution RS maps (e.g. Landsat-7, with overall accuracies ranging from 48% (9 classes) to 77% (4 classes) (Capolsini et al. 2003) and as other LEK maps (65%) (Lauer and Aswani, 2008). Thus both mapping approaches used here have suitable accuracy for conservation applications. High accuracy and high spatial resolution benthic mapping are particularly important for threatened areas, such as the Danajon Bank, because such maps can support the conservation of threatened ecosystems. Having quantified the accuracy and other characteristics of two mapping approaches, I discuss the implications of using each in conservation planning, and effects of the biases and limitations present in each method.

4.4.1 Conservation applications of LEK and RS maps

For conservation planning that considers connectivity and species movement, our research suggests that the best mapping approach will depend on the sensitivity of fish or invertebrate species to seascape patterns. At the individual level, the distribution of habitats, combined with species behavior, can facilitate or impede the movement of organisms (Grober-Dunsmore et al. 2007). For example, individuals traveling across exposed habitats (e.g. sand, deep water) may have a higher predation risk (Selgrath et al 2007) and may avoid crossing open areas (Hovel and Wahle 2010). When suitable patches are far apart their low connectivity can reduce ecological resilience by impeding the movement of mobile organisms with important ecological functions (e.g. roving herbivorous fish) (Nyström and Folke 2001, Olds et al. 2012a). At the population level, connectivity between habitats can enhance the performance of MPAs by supporting higher biomass of piscivores and herbivores than isolated locations (Olds et al. 2012b, but see Edgar et al. 2014). Since RS was better at mapping habitat edges and small patches, it could be most suitable for species with short movement ranges or high predation risk at edges. When important species have wider home ranges or are not impacted by edge effects then both maps have the potential to provide insight into conservation of biodiversity and species.

The map comparison presented here demonstrates that there are several situations where RS maps are most suitable. When conservation programs aim to assess habitat changes due to protection or anthropogenic impacts, consistency is essential for identifying change (Scopélitis et al. 2009, Roelfsema et al. 2013a). Thus the more automated RS approach would be preferred because it offers finer spatial resolution and spatial consistency and is less subjective in comparison to manual interpretation. Additionally, the RS approach is repeatable over short time scales (e.g. one year or less), which may be difficult with the LEK approach (e.g. research fatigue) (Reed 2008). Finer resolution benthic habitat maps (e.g. < 10 m), such as those derived from high spatial resolution imagery, may also be better suited for species with strong habitat associations. Fine-spatial scale habitat variations (5 – 20 m) have been the most important predictors of reef fish community composition (Knudby et al., 2010, but see Mellin et al., 2010). Finally, conservation initiatives that target marine species sensitive to edge effects should incorporate RS maps when possible because the RS approach was much better at capturing habitat edges which can influence species distributions at small-scales (e.g. < 5 m) (Selgrath et al. 2007).

Our findings determined that LEK maps can well represent coarse habitat patterns, in addition to having benefits that extend beyond the maps themselves. In Tanzanian and U.S. Virgin Island coral reefs, some functional groups of fish are more abundant at coral reefs located near seagrass beds (Grober-Dunsmore et al. 2008, Berkström et al. 2013). Such general habitat patterns are at scales coarse enough (e.g. 750 m) to be captured by the LEK-derived maps created here and therefore do not require the more detailed maps provided by high spatial resolution RS. Beyond map characteristics, LEK maps have the benefit of involving communities as active participants in conservation programs, which can lead to the greater success of conservation programs (Reed

2008, Pajaro 2010, Thornton and Scheer 2012). Furthermore, I estimate the LEK approach to cost substantially less than the RS approach, making it a financially practical mapping approach.

4.4.2 Accounting for biases and limitations of maps

Both mapping approaches used here can be improved by accounting for two factors: sampling bias and knowledge/ technological limitations. First, both LEK and RS field survey approaches rely on the observer (resource users or field biologists) identifying the dominant habitat. Observer bias towards a particular resource (e.g. coral) can influence observations and survey results that underlie maps (Roelfsema and Phinn 2008) and LEK mapping may have a systematic offset from scientific observations (Aswani and Lauer 2014). When there are biases in LEK mapping of habitats (e.g. over-mapping 'Rubble' as seen here), RS mapping can create maps with higher accuracy. Alternatively, LEK biases could also be accounted for in mapping procedures (e.g. by using local habitat classifications; Lauer and Aswani, 2008). For RS mapping, field biologists can avail of techniques that minimize observer bias (e.g. standardization between observers; English et al., 1997). When habitats are rare, such as the 'Coral' habitat class in this study, validation surveys could utilize methods for sampling rare species to ensure that there is a high sample of rare habitat classes in the validation survey.

Accounting for the knowledge and/or technological limitations of the mapping approaches used here is a second factor, which can improve map accuracy. Here, I accounted for the limits of fishers' spatial precision by providing respondents with a map that incorporated a satellite image. As fishers' knowledge is limited to places that fishers visit (Lauer & Aswani 2008; Roelfsema et al. 2013a), I addressed this limitation by constraining mapping to fishing grounds. However, this restriction caused areas where fishers do not fish (e.g. mangroves) to be under-mapped. Future projects could address this limitation by asking fishers to map habitats in their fishing grounds and the surrounding areas, or by providing respondents with a grid of data points on a map (Roelfsema et al 2008). Here I found that offshore areas (further from villages) appeared to be mapped less accurately, and such lower accuracies have been found to occur in areas that fishers rarely visit (Lauer and Aswani 2008). In contrast, RS technologies are limited by factors such as the precision of calibration surveys, spectral resolution, water clarity, and water depth (Mumby et al. 1997). By creating a high-accuracy map using a mix of calibration survey methods (with high and moderate precision), I created a RS map with high accuracy. This demonstrates that a practical approach to calibration surveys can yield maps with high value for conservation, even in large areas with variable water clarity.

4.5 Conclusions

Both LEK and RS maps are valuable tools for meeting the growing need for improved marine habitat maps. By creating detailed maps of benthic habitats for the Danajon Bank coral reef, I developed valuable assets for species conservation and spatial planning in the center of marine biodiversity and identified guidelines for future conservation mapping. I suggest that programs carefully consider the specific goals and uses for maps, as well as the resources available, when deciding upon the most suitable approach. For projects requiring high spatial precision or high habitat accuracy, the RS method would be the best option. When resources are limited or objectives dictate it, LEK mapping can provide a viable alternative to RS and has the added benefit of engaging stakeholders. There is also the possibility of enhancing both maps by combining them into one (e.g. LEK in nearshore or deeper areas with turbid water and RS in shallow offshore areas with clear water; Ban et al., 2009; Roelfsema et al., 2009). Drawing on the strengths of each approach has the potential to improve conservation efforts in ways that range from confidence in reserve design (Tulloch et al. 2013) to more accurate evaluations of restoration and conservation targets (Gergel et al. 2007). Overall, both mapping approaches, apart or together, have the potential to aid informed decision-making to achieve conservation targets.

Chapter 5: Synergistic stressors offset by depth, management, and landscape structure in a coral reef ecosystem

5.1 Summary

The influence of multiple stressors on ecosystems has proven difficult to quantify and presents a key challenge for conservation. In a coral reef ecosystem in the central Philippines, I used a distinctive combination of satellite imagery and participatory mapping to gather social-ecological data with a high spatial resolution. I then identified anthropogenic and biophysical characteristics of the system that were correlated with the presence of living coral. Several stressors had negative, additive effects on the probability that a location supporting living coral: total fishing pressure from all small-scale fishing gears (88 gears), blast fishing pressure, and human population density. Cumulative fishing effort and population density had a negative, synergistic effect on the probability that an area supported living corals, while blast fishing and population density had an antagonistic effect. I found that the probability that an area supported living corals in 2010 was negatively related to cumulative fishing pressure from the past 10-30 years (1980-2000). Thus the relationship between the probability that an area supported living corals and past fishing pressure exhibited a lag. Variables associated with a higher probability of a location supporting living corals included marine reserve protection and increasing depth. Overall, the strongest influence on the probability that a location supported living coral was the spatial arrangement of habitats. Specifically, living corals were more likely to be present in compact habitat patches than in fragmented habitat patches. These relationships offer guidance for conservation including managing ecosystems to optimize beneficial landscape characteristics, reducing the spatial overlap of synergistic stressors, and ensuring that resource users recognize

and mitigate stressors that accumulate slowly, but cause significant disturbances to ecosystems. By successfully integrating local environmental knowledge and satellite-based mapping, I demonstrate an opportunity for monitoring the effects of stressors on coral reefs at the scale of ecosystems.

5.2 Introduction

Conservation today must navigate complex priorities because anthropogenic stressors permeate all ecosystems, including protected areas (Steffen et al. 2011). This pervasive influence of human activity has led to a call by scientists and policy-makers to augment the benefits of protected areas by improving ecosystem functioning in human-dominated landscapes. Recent work on a global scale has approached this problem by identifying characteristics of regions containing bright spots – places that fare better than expected under stress (Cinner et al. 2016). However, questions remain about reducing the impacts of stressors within ecosystems due to unknown interactions between overlapping stressors and the inherent biophysical variability of ecosystems (Crain et al. 2008). One promising opportunity to explore these questions is to characterize places that have withstood or adapted to existing disturbances and to develop conservation and management programs that foster beneficial characteristics, while strategically reducing stressors. However, until recently, the information needed to take this opportunity has been limited, particularly for marine ecosystems.

The role of stressors in shaping ecosystems is widely recognized, yet the effects of individual stressors, their legacies, and their interactions remain poorly understood (Halpern et al. 2015, Côté et al. 2016). Stressors such as over-exploitation, pollution, and habitat loss have demonstrated effects on ecosystems, including altering ecological dynamics and threatening important species (Underwood 1989, Estes et al. 2011, McCauley et al. 2015). Their effects may appear quickly or may accumulate gradually over time, and effects may have a positive or negative influence on ecosystem conditions. Where multiple stressors affect an area, stressors primarily interact in three ways: additive, antagonistic (less than additive), or synergistic (greater than additive) (Crain et al. 2008). The possibility of synergistic effects are particularly concerning because they have the potential to unexpectedly accelerate environmental change (Lindenmayer et al. 2010). When ecosystems contain locations with distinct stressor combinations, this spatial variability can support evaluations of individual and overlapping stressors.

Despite clear benefits from understanding how stressors influence ecosystems, spatial datasets documenting ecosystem-scale stressors over multiple decades are uncommon (Geldmann et al. 2014). These datasets are rare for several reasons including the difficulties of large scale experiments, and insufficient monitoring of existing stressors – particularly in developing countries (Crain et al. 2008). Where data about stressors is unavailable, surrogates for stressors can include proxies, anecdotal information, and local environmental knowledge (LEK) (Thurstan et al. 2015). Using surrogates to quantify the historic distribution and intensity of stressors can provide context in ecosystems where past disturbances continue to shape today’s conditions (McKey et al. 2010, Tomscha and Gergel 2016). Moreover, long-term data can be necessary to identify stressors that have delayed effects (Scheffer et al. 2001, Walker and Salt 2006).

One characteristic of ecosystems that can influence their ability to absorb disturbances is the spatial arrangement of habitats at broad and local scales. At the broadest scale, 'landscapes' or 'seascapes' are heterogeneous geographic areas supporting a diversity of habitats. The spatial arrangement of habitats influences the landscape's connectivity – the movement of organisms and materials through ecosystems (Calabrese and Fagan 2004, Wedding et al. 2011, Turner and Gardner 2015). In stressed landscapes, connectivity affects the rate at which propagules (e.g. coral larvae, seeds) and adults colonize disturbed areas (Nyström and Folke 2001, Dethier et al. 2003, Smith et al. 2011). Another aspect of landscape connectivity involves the distance between different habitats, for example between corals and mangroves (Olds et al. 2012c). Large distances between critical habitats can impact species that use multiple habitats in a day, season, or during successive ontological stages (Werner and Gilliam 1984, Davis et al. 2014).

By influencing ecological functioning, critical habitats, such as living corals, enhance the capacity of ecosystems to withstand disturbance (Holling 1973, Walker and Salt 2006). When considering habitats at local scales (tens of meters), characteristics of patches – continuous areas containing one relatively similar habitat type – can affect ecological dynamics such as predation and competition (Andren and Angelstam 1988, Selgrath et al. 2007, Hovel and Wahle 2010). Influential characteristics of patches include their area size, length of edges, and structural complexity (Turner and Gardner 2015). In coral reefs, for example, living corals provide complex habitat structure, which offers shelter for several species (Almany 2004). In contrast, dead corals erode, which simplifies coral structure and limits their function as a shelter for other animals (Alvarez-Filip et al. 2009, Graham and Nash 2013). Due to the structural differences between living and dead corals, reefs that are dominated by living coral support a greater abundance and biomass of marine life (Pratchett et al. 2014). Coral-dominated reefs are also characterized by a higher biodiversity, and which provides these reefs with greater functional and response diversity (Sanciangco et al. 2013).

In this study, I characterize locations within a coral reef ecosystem that have withstood or adapted to local stressors as indicated by the dominance of living coral. Corals are particularly good for exploring long-term ecosystem status because when corals are stressed and die, physical evidence of their prior distribution can still be documented. Furthermore, corals are a highly threatened group of species that are foundational for coral reef ecosystems. I ask three core questions: (1) How is the spatial distribution of living coral related to anthropogenic stressors and biophysical conditions? (2) Do multiple stressors affecting corals have additive, synergistic, or antagonistic impacts? (3) Over what time-scale do corals respond to fishing pressure (1960-2010)? For this analysis I combine high spatial resolution satellite imagery and maps depicting local environmental knowledge (LEK) to assess coral, ecological conditions, landscape characteristics, and anthropogenic influences.

5.3 Methods

5.3.1 Study site

The Philippines, located in the global center of marine biodiversity, supports 21,983 km^2 of coral reefs and is a global priority for conservation (Roberts et al. 2002, Carpenter and Springer 2005, Halpern et al. 2008, Sanciangco et al. 2013). The Danajon Bank (Figure 5.1; 10˚15’0’N, 124˚8’0’E) is one of only six double barrier reefs in the world and sits off the northern edge of Bohol – a province characterized by small-scale farming, deforested scrubland, and small-scale mining, as well as high levels of erosion (Provincial Government of Bohol 2011). Several characteristics of the Danajon Bank potentially enable it to withstand anthropogenic stressors, including high biodiversity, relatively high aragonite saturation, and low incidence of both coral disease and coral bleaching (Roberts et al. 2002, Cao and Caldeira 2008, Burke et al. 2011, McClanahan et al. 2012, Philippine Coral Bleaching Watch 2013), although data for many of these stressors are currently sparse or at coarse scales. Currents primarily flow alongshore, and switch directions seasonally due to seasonal weather patterns (Villanoy et al. 2006). Overall the area is characterized by a well-mixed water column, weak tidal circulation, and the absence of large waves (Villanoy et al. 2006). In the study area, the largest current exchange is in the Northwest Pass (Figure 1) where several channels converge (Villanoy et al. 2006).

The Danajon Bank lies very close to Cebu City (the second largest metropolitan area in the Philippines, 2.8 million people), but is itself located in a rural region, which struggles with extreme poverty and minimal infrastructure. Between 45% and 70% of the residents live below the Philippines poverty level (Provincial Government of Bohol 2011) and island villages generally lack electricity, running water, and septic systems. Small-scale ('municipal') fisheries have exclusive rights to fish in the Danajon Bank. These fisheries use many types of fishing gears and target diverse species of fish and marine invertebrates (Green et al. 2000, Selgrath et al. 2017a). Since 1998, small-scale fisheries have been co-managed by local governments and resource users, with varying levels of local engagement. During the same period, approximately 70 communities have established locally-enforced no-take marine protected areas (MPAs), including 12 MPAs in our study area. Despite these management efforts, heavy fishing pressure,

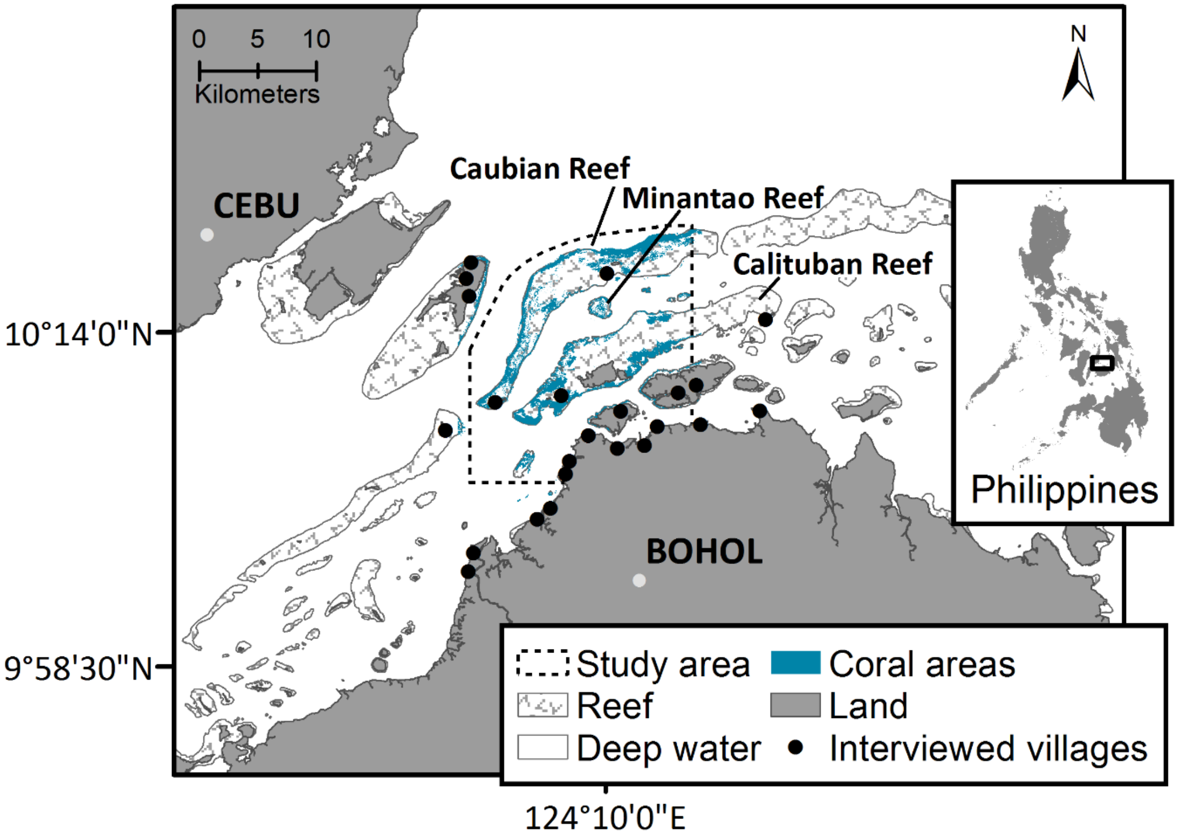

Figure 5.1 Study site in the Danajon Bank, Central Visayas, Philippines. Blue indicates locations with coral or rubble within the study area.

destructive fishing, and high population densities have depleted marine life and degraded much of the marine environment (Christie et al. 2006, Marcus et al. 2007). Yet there are still some places that support living corals (Selgrath et al. 2016) (Figure 5.5.2). This pattern suggests spatial variations in the capacity of this ecosystem to withstand stress (Nyström and Folke 2001, Olds et al. 2012c, Selgrath et al. 2017b). I focused on a 19 km by 22 km area in the central Danajon Bank (Figure 5.5.1).

5.3.2 Overview

I evaluated the relationship between reef state (presence/absence of living coral or rubble), (Figure 5.5.2) and 25 spatially-explicit variables describing anthropogenic stressors and biophysical attributes about the reef system (Table 5.1). These social-ecological attributes were derived from remote sensing imagery and from a participatory mapping exercise with local fishers (Figure 5.3), as well as from publically available data sources (e.g. Philippines census data (National Statistical Coordination Board 2010); data layers provided by Reefs at Risk Revisited (Burke et al. 2011)). I considered additive, antagonistic, and synergistic effects of stressors to understand their influence on the system using hierarchical models. Next, I describe each of these components in greater detail.

For this analysis, I used a binary measurement for reef state with reefs classified as (a) 'coral-dominated', or (b) 'rubble dominated.' The rubble-dominated category included degraded rubble, shattered coral, deal coral, and macroalgae. Since the Danajon Bank contains areas with mixed habitats, the rubble-dominated class included areas containing rubble mixed with other habitats (e.g. sand, seagrass). During map classification, I used an approximately 30% cover threshold to consider reefs to be dominated by living coral (see 5.3.3.2.1 Marine Habitats below for classification details). Using a binary variable as a proxy for reef functioning represents the inherent trade-offs between highly detailed surveys of small areas and the coarser mapping of ecosystems.

5.3.3 Spatial datasets

5.3.3.1 Fishing and other anthropogenic pressures

Fishing is considered to be one of the greatest threats to coral reef ecosystems (Burke et al. 2011). Therefore, I developed fishing maps during participatory mapping interviews with 391 randomly sampled fishers, 295 of whom fished in the mapped area. Fishers were selected using census data routinely collected by village health care workers in the Philippines. Interviews occurred in 23 communities (villages and towns) between July 2010 and April 2011. I focused on men's fisheries, and did not interview women, whose fisheries are dominated by gleaning in intertidal areas (Kleiber et al. 2014). During interviews, I mapped the spatial and temporal distribution of respondents' fishing effort over their career as fishers. A detailed description of interview methods is available in Appendix A. I included effort from 88 fishing gears (e.g. hand-line, bottom-set gillnets, skin diving). Based on interviews, I estimated that during 2010 approximately 8,000 men fished inside the 418 km^2 study area.

Table 5.1 Variables used in modeling. Variables included in final model are shown in bold.

Category	Variable	Method/Description	Resolution	Source	Reference
Response	*State of corals*[1]	*> 30% coral cover*	*2 m*	Selgrath et al. 2016	Gomez et al. 1994, Bruno and Selig 2007
		> 30% rubble, dead coral, or macroalgae, except where coral cover > 30%	*2 m*	Selgrath et al. 2016	
Ecological	*Depth*[1]	*Depth of reef*	*10 m*	Modeled based on depth point readings digitized from NAMRIA maps by the FISH Project	Graham et al. 2015
	Distance from rivers	Distance to river mouths. Proxy for sedimentation.	2 m	Modeled based on WorldView2 Satellite images	Hansen et al. 2011
	Seagrass isolation[1]	*Distance between coral or rubble patch and seagrass patch*	*10 m*	Modelled based on maps from Selgrath et al. 2016	Wedding et al. 2011
	Mangrove isolation	Distance between coral or rubble patch and nearest mangrove patch	10 m	Modelled based on maps from Selgrath et al. 2016	Wedding et al. 2011
	Coral isolation	Distance between coral or rubble patch and nearest coral patch	10 m	Modelled based on maps from Selgrath et al. 2016	Wedding et al. 2011
	Patch compactness[1]	*Patch compactness, accounting for patch area (-1 when the patch is maximally compact, decreases as the shape becomes irregular); Inverse of Patch Shape*	20 m	Modelled based on maps from Selgrath et al. 2016	McGarigal et al. 2012
	Patch area	Area of the coral or rubble patch	2 m	Modelled based on maps from Selgrath et al. 2016	McGarigal et al. 2012
	Patch edge length	Length of the edge of coral or rubble patches	2 m	Modelled based on maps from Selgrath et al. 2016	McGarigal et al. 2012

Category	Variable	Method/Description	Resolution	Source	Reference
	Patch near-neighbor distance	Distance between a patch and the nearest patch of the same habitat	20 m	Modelled based on maps from Selgrath et al. 2016	McGarigal et al. 2012
	Past thermal stress (1998-2007) [2]	Severe thermal stress defined as a NOAA Bleaching Alert level > 2 (DHW $\geq$ 8) at least once between 1998 and 2007 or an observation of severe coral bleaching from ReefBase (1998-2007) or from the Philippine Coral Bleaching Watch.	1 km	Data from World Resources Institute incorporated bleaching observations (1998-2007) from ReefBase with UNEP-WCMC Bleaching Data, WorldFish Center, www.reefbase.org, 2009; and satellite-detected thermal stress (1998-2007) from National Oceanic and Atmospheric Administration, Coral Reef Watch, Degree Heating Weeks data (calculated from NOAA's National Oceanographic Data Center Pathfinder Version 5.0 SST dataset), http://coralreefwatch.noaa.gov, 2010. WRI data were supplemented with data from Philippine Coral Bleaching Watch.	Burke et al. 2011
	Aragonite[2]	Aragonite levels	200 km	Adapted from Cao and Caldeira 2008 for use in the Reefs at Risk Revisited project.	Burke et al. 2011
	Coral disease[2]	Observations of coral disease	Point data	Data from Philippine Coral Bleaching Watch. Note that few observations existed for the Danajon Bank.	Burke et al. 2011
Anthropogenic	*MPA*[1]	*Binary variable indicating if a location is located in a protected area*	20 m	Integrated data from ZSL Philippines MPA database and	Halpern and Warner 2002

Category	Variable	Method/Description	Resolution	Source	Reference
				coral triangle atlas (http://ctatlas.reefbase.org/)	
	Contemporary fishing effort (2010)[3,5]	Estimated yearly fishing pressure (fished days per year) by all fishers	20 m	Modeled based on fishing maps from Selgrath et al. 2017	Halpern et al. 2008
	Cum. fishing effort (1980 - 2010)[3,4]	Maps of cumulative fishing effort in 10 year time intervals	20 m	Modeled based on fishing maps from Selgrath et al. 2017	Halpern et al. 2015
	Fishing legacy (1980-2000)[1,3]	*Sum of fishing effort from 1980-2000 in 10 year time-steps*	20 m	Modeled based on fishing maps from Selgrath et al. 2017	Halpern et al. 2015
	Fishing change (2000-2010)[3]	Change in fishing effort from 2000 to 2010	20 m	Modeled based on fishing maps from Selgrath et al. 2017	Halpern et al. 2015
	Blast fishing (2000-2010)	*Estimated levels of blast fishing for 2000 and 2010*	20 m	Modeled based on fishing maps from Selgrath et al. 2017	Alcala and Gomez 1987
	Population density risk[1,3]	*Undocumented stressors to the ocean (e.g. nutrient loading, trampling) are influenced by the population density of adjacent communities*	*10 m*	*Modelled based on inverse distance to villages. Village populations derived from village census data.*	McPherson et al. 2008
	Market proximity	Distance from the regional Pasil fish market in Cebu City	10 m	Modelled based on market location.	Brewer et al. 2012
	Town proximity	Distance from towns (enforcement centers, weekly local markets)	10 m	Modelled based on town locations.	Brewer et al. 2012
	Community proximity	Distance to fishing villages and towns (fishers, enforce MPAs)	10 m	Modelled based on village locations.	Ban et al. 2009b, Brewer et al. 2012
Random Effects	*Ecological zone[1]*	*Ecological conditions in Danajon Bank: inshore turbid, mid, offshore clear*	*20 m*	*Manually delineated based on Hansen et al. 2011 and WorldView2 satellite images.*	Hansen et al. 2011

Category	Variable	Method/Description	Resolution	Source	Reference
	Geomorphology	Reef flat, reef crest, reef slope, etc.	2 m	Delineated from WorldView2 satellite images.	Roelfsema et al. 2013, Selgrath et al. 2016

1. Part of final model, 2. No variation within study area, 3. Normalized by dividing value in each pixel by the maximum value. For normalizing fishing I used the maximum value for all years (from 2000). 4. I assessed 50 years of cumulative impacts (1960 – 2010), but the additional years (1970, 1960) were not informative. 5. Fishing effort was estimated for men fishing using any gears and for destructive gears only. No metrics using destructive gears were significant.

I created maps of fishing pressure using 10 m x 10 m pixels (Figure 5.3) at decadal intervals from 1960 to 2010. I mapped the total effort over one year (days per year fished for all fishers combined) for each pixel in the study area by overlaying the maps of each individual's fishing activities. To compare relative fishing pressure across space and time, I normalized fishing effort (from 0-1) for each pixel relative to the maximum of all years (see Halpern et al. 2015, Maynard et al. 2015 for further discussion on normalizing variables). Based on the maps quantifying total fishing effort in one year, I produced four measures of fishing pressure: (a) contemporary fishing pressure (2010); (b) cumulative fishing pressure (the sum of current and past fishing pressure); (c) lag fishing pressure (the sum of past fishing pressure, excluding 2010 fishing pressure to account for time-lag effects); and (d) change in fishing pressure (2010 fishing pressure minus 2000 fishing pressure). Lag responses occur when the influence of a stressor from the past does not become apparent for a period of time. For the (b) legacy and (c) lag measures, I evaluated iterations covering different time spans (e.g. 2000-2010 vs. 1970-2010) (Table 5.2). Finally, I created similar fishing pressure maps for fishing gears classified as destructive (e.g. blast fishing using explosives, small-mesh nests, etc.). Since some destructive gears may not affect coral directly, I also evaluated blast fishing independently. Blast fishing is an extremely destructive method that often destroys underlying habitats, including corals.

Human population and market data. I used population information collected from village census data to create four variables: population density, distance to the regional fish market in Cebu City, distance to towns, and distance to fishing communities (Figure 5.3). Several ocean stressors (e.g. nutrient loading and trampling) are correlated with the population densities of adjacent communities (Mora, 2008). I assumed population density impacts affected reefs up to 1.5 km from populations (McPherson et al. 2008). Therefore, I used distance decay with a square root decay to calculate the risk from population density that decreased with distance to the communities (McPherson et al. 2008). I normalized the population densities by dividing the values in each pixel by the highest value estimated. Finally I evaluated the influence of protection using a map of existing protected areas.

5.3.3.2 Biophysical attributes

5.3.3.2.1 Marine habitats

I created maps of marine habitats by classifying high spatial resolution (2 m) multi-spectral satellite images (WorldView2, dates: 2010/05/10 and 2012/20/04) using object-based image analysis (OBIA). These two dates were the earliest times with clear WorldView2 images of the study area, as the satellite was launched in October 2009. OBIA involved image segmentation and classification using field-verified training and testing data. Training data were collected through georeferenced point intercept transects and geo-referenced spot-check surveys, by placing a viewing bucket in the water to estimate the percent cover benthic cover types (Figure H.1). Verification data were collected via in-water georeferenced surveys using a mix of point-intercept transects and bucket viewing (surveys conducted 2007-2009). These georeferenced ground verification points were used to train OBIA decision rules that

Table 5.2 Four measures of fishing pressure were used in modeling: Contemporary, Legacy, Lag, and Difference.

Measure of fishing pressure	Year Range	Description	Year(s)					
			1960	1970	1980	1990	2000	2010
Contemporary	2010	Single year						+
Legacy	2000-2010	Added pressure from 2 years					+	+
Legacy	1990-2010	Added pressure from 3 years				+	+	+
Legacy	1980-2010	Added pressure from 4 years			+	+	+	+
Legacy	1970-2010	Added pressure from 5 years		+	+	+	+	+
Legacy	1960-2010	Added pressure from 6 years	+	+	+	+	+	+
Lag	2000	Single year					+	
Lag	1990-2000	Added pressure from 2 years				+	+	
Lag	1980-2000	Added pressure from 3 years			+	+	+	
Lag	1970-2000	Added pressure from 4 years		+	+	+	+	
Lag	1960-2000	Added pressure from 5 years	+	+	+	+	+	
Difference	2010-2000	Subtracted pressure from 2 years					-	-

classified the satellite images using color, texture, and geographic location (See Roelfsema et al. 2010 and Selgrath et al. 2016b for further details on methods) achieving over 75% accuracy, a level of accuracy which is 125% higher than the standard for maps of marine habitats (standard: 60% accuracy) (Roelfsema and Phinn 2013). Mapping was restricted to shallow areas (approximately 15 m depth) because of limitations of deep-water image classification. Geomorphic zones (e.g. reef crest, reef flat) were identified and then further classified into 16 benthic (seafloor) habitat types (or classes). For this analysis, I simplified habitats into 5 classes: (living) coral, rubble, sand, seagrass, and mangroves. As the OBIA map did not capture seagrass beds in turbid, shallow waters, I supplemented the OBIA map. I included seagrass beds from a habitat map created through interviews with fishers. See Chapters 3 and 4 for additional details on mapping methods.

5.3.3.2.2 Habitat configuration

From the habitat map I derived landscape pattern indices describing the arrangement of habitats in the seascape (Table 5.1). I primarily emphasized class-level indices describing coral and rubble patches (e.g. mean patch size, distance between patches; Figure 5.2) as well as their relationships to other habitat classes (Table 5.1, Figure 5.3). Nearest neighbor distance was used to quantify distance between coral (or rubble) and the nearest patch of similar habitat. To characterize specific habitat patches, I estimated area, the length of edge, and patch compactness (incorporating edge: area ratio and patch size; inverse of patch shape as per McGarigal et al. 2012. The metric for patch compactness increases to -1 when maximally compact (i.e., square or nearly so) and decreases without limit as the shape becomes more irregular. Because mangrove and seagrass habitats influence reef fish populations and corals (Grober-Dunsmore et al. 2009, Olds et al. 2012c), I also determined the isolation of reefs from these two habitats. I calculated all landscape indices in R (R Core Team 2016), ArcGIS 10.4 (Environmental Systems Research Institute, Redlands, CA), and Fragstats v. 4 (McGarigal et al. 2012).

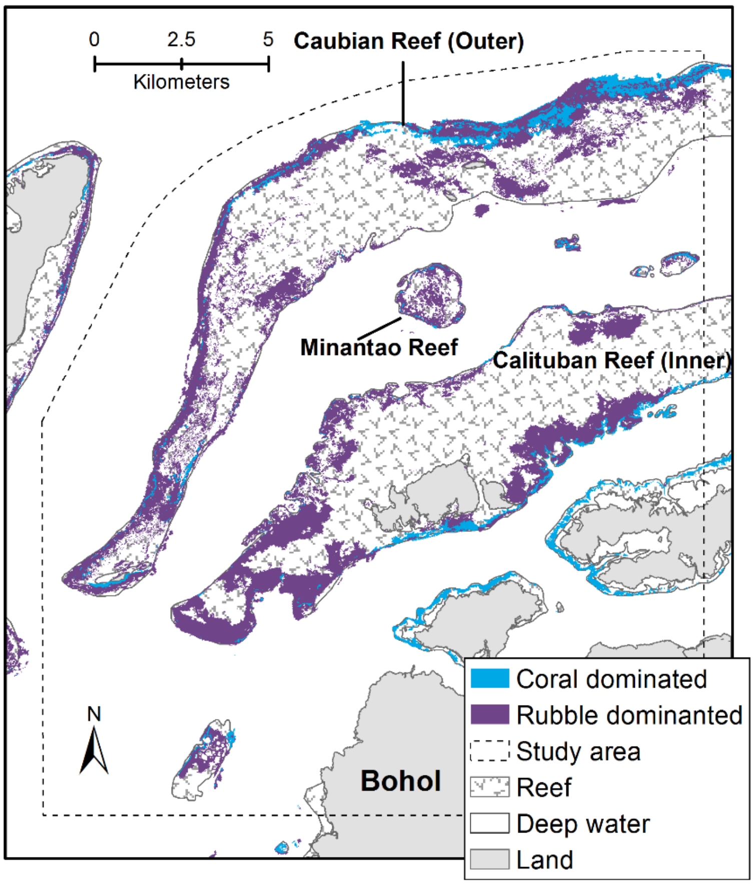

Figure 5.2 Map of coral and rubble locations in the study area, Danajon Bank, Philippines.

I obtained information for several additional biophysical attributes using existing sources (Table 5.1). First, I created a bathymetry layer by interpolating depth soundings from a NAMRIA nautical chart (Figure 5.3) using a spline with barriers function in ArcGIS 10.4. Second, river mouths were identified using the WorldView-2 satellite images and I classified the seascape based on distance to these rivers. Third, I classified the study area into three ecological zones following Hansen *et al.*, 2011: an coastal zone of turbid waters, extensive mangroves, and terrestrial islands that support farming (inshore); an offshore, outer reef zone with clear waters, extensive seagrass beds, scattered cays; and an inner reef zone with intermediate characteristics of the coastal and outer reefs. Fourth, I used a map of existing marine reserves to examine the influence of protection on reef state (Figure 5.3). Fifth, I used data from regional models of aragonite saturation and past thermal stress (1998-2007) (Burke et al. 2011). Other variables that may influence the presence of living corals (e.g. water clarity, *chl a*) were not available for this region.

5.3.4 Statistical analysis

5.3.4.1 How is the spatial distribution of living coral related to biophysical conditions and anthropogenic stressors?

I evaluated the relationship between reef state (living coral or rubble), ecological variables, and anthropogenic influences. To identify and exclude highly correlated variables ($r > 0.7$), I used Pearson correlation coefficients. When variables were correlated, I included the variable that most directly measured a phenomenon (e.g. fishing effort vs. distance to markets) or which I expected to be more informative based on published research. From the maps described above, I sampled 3,232 random points at locations in the study area. To identify the best model, I used forward stepping logistic regressions with hierarchical modeling and compared models based on AIC and the distribution of residuals (Gelman and Hill 2007). I used forward stepping models because the model with all variables would not converge. I iteratively started the stepping process with different variables, and started the final stepping procedure using variables, which were consistently significant in all models. It is currently not possible to include a spatial correlation structure in generalized hierarchical models (for a discussion, see http://bit.ly/2hSMnV7). Thus I sampled points separated by a minimum distance of 100 m to reduce potential spatial autocorrelation. I also looked for spatial structure in model residuals (details below). Final variables were ranked in influence using the Wald Test. For a random

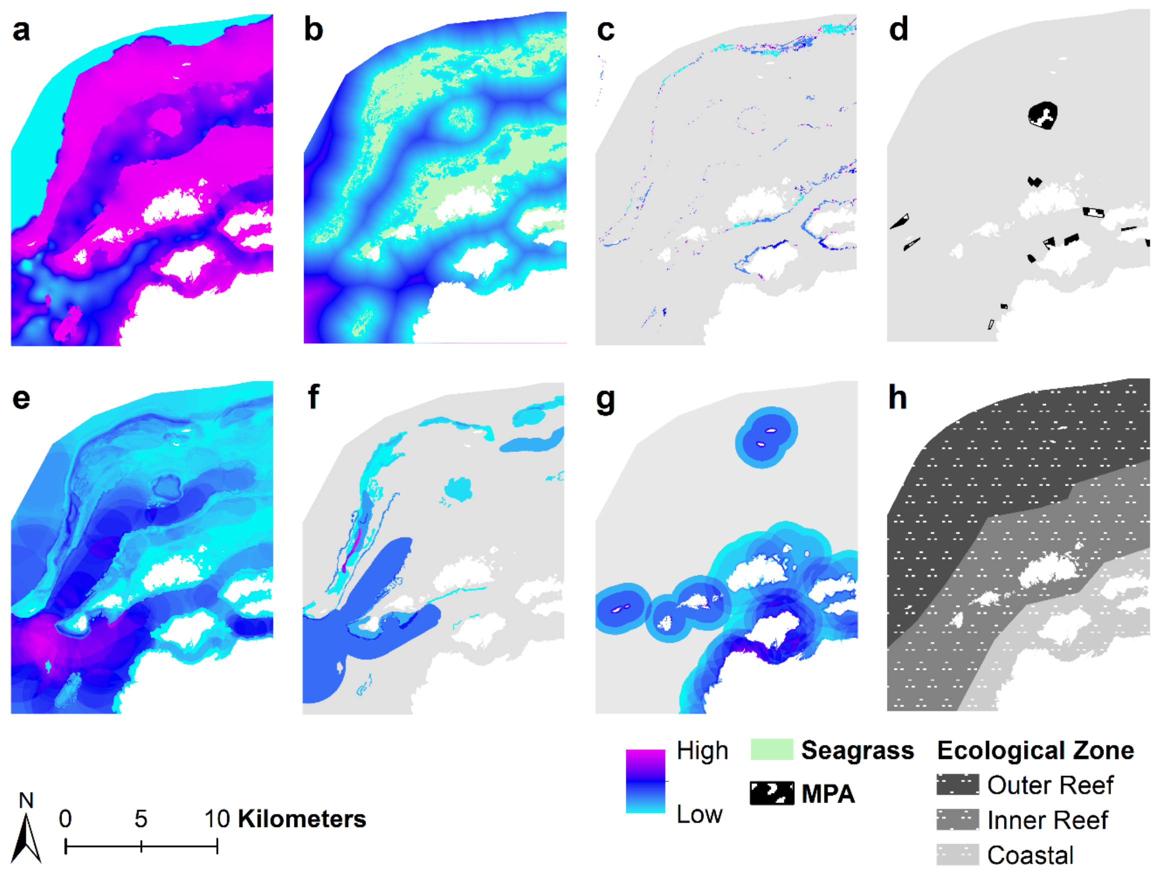

Figure 5.3 Independent variables that were significant for modelling where living coral was present in the Danajon Bank, Philippines included (a – f) fixed effects and (g) random effects: (a) Depth; (b) Seagrass isolation; (c) Patch compactness; (d) MPA locations; (e) Fishing legacy (1980-2000); (f) Blast fishing (2000-2010); (g) Population density; (h) Ecological zone.

effect, I used ecological zones. Initially I included geomorphology (e.g. reef crest, ref slope) as second random effect, but this variable did not significantly improve the models (p = 1.0). I conducted analyses in R 3.3.1 (packages: lme4, car) (Fox and Weisberg 2011, Douglas Bates et al. 2015, R Core Team 2016).

Several variables of interest were also excluded because they did not vary meaningfully over the study area, including aragonite saturation, past thermal stress (1998-2007), and presence of coral disease (Table 5.1). Finally I log transformed and standardized all variables excluding those that were binary or categorical. Data are from 2007–2012, except for fishing data, which extend from 1960-2010.

5.3.4.2 Over what time-scale do corals respond to fishing pressure?

To identify the most informative time-scale for fishing pressure, I compared the relationship between reef state and four quantifications of fishing pressure (contemporary, cumulative, lagged, change) (Table 5.2). I substituted measures of fishing pressure into the model and identified informative time-scales based on the Akaike information criterion and the Wald Test. For these comparisons I considered all fishing methods, destructive fishing methods, and blast fishing. However, I did not include correlated fishing values in the same model.

5.3.4.3 Do multiple stressors affecting corals have additive, synergistic, or antagonistic impacts?

To identify additive and synergistic effects, I assessed models with interactions among anthropogenic stressors. I was, however, unable to consider interactions among all stressors because some variables were correlated.

5.3.4.4 Integrating the three questions

I integrated the three research questions using three steps: evaluating spatial patterns in model residuals, testing the model's predictive ability, and evaluating the effectiveness of a reduced model. Using the final model, I assessed spatial patterns in the model's residuals. First I mapped the distribution of model outliers and used outlier locations to visually assess characteristics of bright spots. Second, I evaluated changes in the distributional characteristics of outliers using geographical weighting. I calculated and mapped geographically weighted summary statistics for the absolute value of model residuals (R package: GWmodel; moving window bandwidth: 1 km). To test predictive ability, I evaluated the power of the best model to estimate the probability of coral at 471 independent points randomly sampled from the habitat map. Testing points were a minimum of 50 m from training points and at least 100 m apart. Currently, landscape pattern indices are uncommon in existing coral reef monitoring programs and few coral reef ecosystems have been characterized by high spatial resolution maps. Therefore, I evaluated the predictive power of a reduced model that included anthropogenic drivers and depth, but excluded landscape pattern indices.

5.4 Results

5.4.1 How is the spatial distribution of living coral related to biophysical conditions and anthropogenic stressors?

The presence of living coral in the Danajon Bank was correlated with both anthropogenic and biophysical variables. I found evidence for the influence of synergies, stressor legacies, management interventions, and landscape patterns. Overall, sites dominated by living coral comprised 15% of sampling points in reef areas, making rubble 5.5 times more common than living coral in our study area.

The probability of an area supporting living corals corresponded to biophysical and management conditions including depth, landscape pattern indices, and protected areas. The probability of an area supporting living corals increased linearly with increasing depth ($p < 0.001$; Figure 5.4). There was a higher probability of living corals being present in locations not adjacent to seagrass patches ($p < 0.001$; Figure 5.4) and in locations where coral or rubble patches were compact (i.e. high compactness values; $p < 0.001$; Figure 5.4). Locations inside protected areas had a higher probability of supporting living coral ($p < 0.01$; Figure 5.4).

5.4.2 Over what time-scale do corals respond to fishing pressure?

Fishing pressure had a negative influence on living corals. When I evaluated fishing pressure at various time scales, the most informative measure was cumulative fishing from 1980 – 2000 (i.e. cumulative fishing pressure with a 10 year lag) (Figure 5.4; $p < 0.01$). Cumulative fishing pressure for all years was also informative ($p = 0.05$), but significantly less powerful than fishing pressure that incorporated a time-lag ($p < 0.001$). Neither contemporary fishing pressure (2010 pressure only), nor the change in fishing pressure (2010 pressure minus 2000 pressure) had predictive power for differentiating living coral and rubble areas ($p = 0.76$ and $p = 0.88$, respectively). Meanwhile, destructive fishing pressure from all destructive gears did not exhibit a significant relationship with living coral, regardless of the time period considered: contemporary (2010: $p=0.24$), historic (1980-2010: $p = 0.82$), lag (1980-2000: $p = 0.65$); or changes (2010 minus 2000: $p = 0.54$). In contrast, higher levels of pressure by blast fishing reduced the probability of an area supporting living corals using contemporary (2010; $p < 0.001$) or cumulative measures, with cumulative pressure from blast fishing over the past ten years having the strongest influence on contemporary corals (2000-2010: $p < 0.001$).

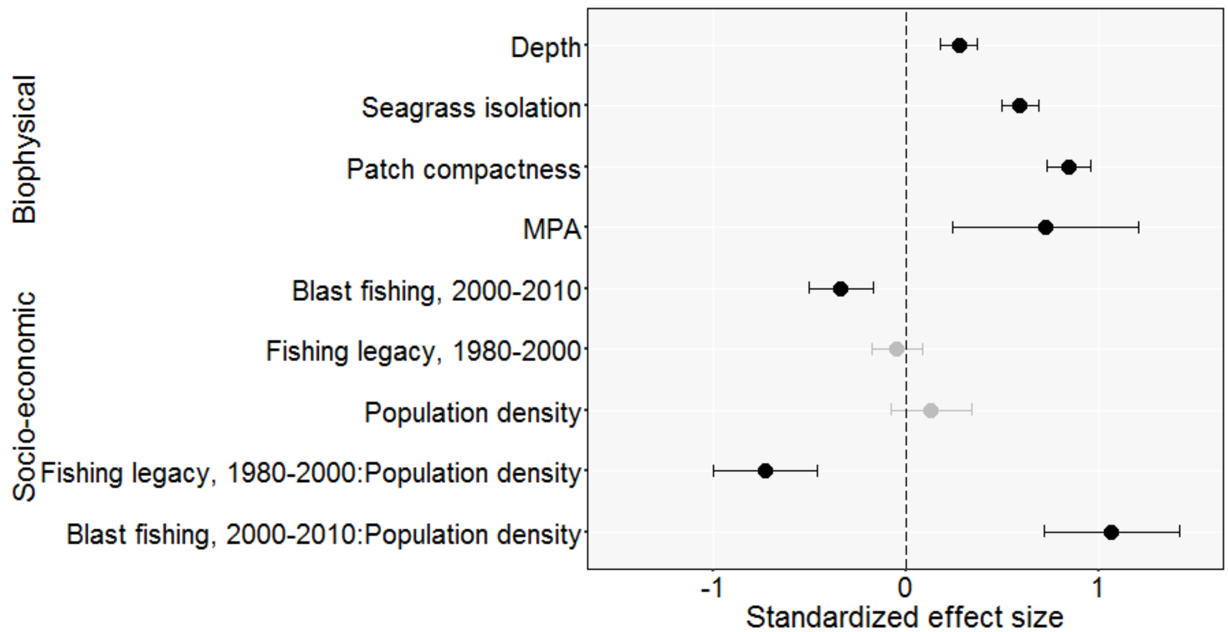

Figure 5.4 Standardized effect sizes for four models predicting the probability of an area supporting living corals being present at locations throughout the Danajon Bank, Philippines. Models varied in their incorporation of stressors. The best model included a decade of blast fishing pressure, the legacy of all fishing pressure (with a lag), population density, and the interaction of the two fishing stressors with population density. Parameter estimates are from hierarchical logistic regression models with 95% confidence intervals. The grey points were not significant in model.

5.4.3 Do multiple stressors affecting corals have additive, synergistic, or antagonistic impacts?

Fishing legacies and population density were synergistic stressors. Living coral was less likely to be present at locations where heavy fishing occurred in close proximity to communities with high population densities (Figure 5.4; $p < 0.001$). In contrast, blast fishing and population pressure exhibited an antagonistic relationship on corals, making corals more likely to be present in areas that were blast fished if those fishing grounds were also close to areas with high human population density (Figure 5.4; $p < 0.001$).

5.4.4 Integrating the three questions

After identifying the best model, I used the Wald test to evaluate the relative influence of the significant predictors. I found compactness of habitat patch to be the most influential variable and fishing legacy to be the least influential predictor of living coral (Table 5.3). I mapped the spatial distribution of model residuals and found no clear spatial patterns, such as directional trends (Figure 5.5a). Bright spots – where live coral existed more frequently than I would have predicted – occurred in 2.7% of the samples (Figure 5.5b). In contrast, 0.2% of the samples were dark spots, locations where live coral existed less frequently than I would have predicted.

Next, I evaluated the predictive power of the final model using independent samples and found that the final model was successful at separating living coral and rubble-dominated areas (Figure 5.7a).

Table 5.3 Wald Chi-squared values for evaluating the relative influence of the various factors influencing the probability of living coral being present at locations in the Danajon Bank.

Variable	Wald chi-sq	Pr > chi-sq
Patch compactness	216.16	<0.001
Seagrass isolation	155.18	<0.001
Population density:Blast fishing, 2000-2010	36.01	<0.001
Depth	32.71	<0.001
Fishing legacy, 1980-2000:Population density	28.12	<0.001
Blast fishing, 2000-2010	17.28	<0.001
Population density	16.98	<0.001
MPA	8.62	<0.01
Fishing legacy, 1980-2000	1.37	0.24

Using geographic weighting, I found that there were no significant spatial patterns in outliers (i.e. all geographically weighted residuals were < 2 standard deviations from 0) (Figure 5.7).

Finally, I assessed the predictive power of anthropogenic drivers and depth alone. Using the reduced variables, I was able to identify a statistically significant model using the main dataset (Figure H.2). However, using independent data to test the reduced model revealed that it had no predictive power and was unable to distinguish between locations with living coral and rubble (Figure 5.7b).

5.5 Discussion

Discriminating among the independent and cumulative impacts of different stressors is an enduring challenge. This is especially so with stressors that are difficult to test experimentally, such as long term fishing pressure (Pendleton et al. 2016), and that vary concurrently over space and time. I provide evidence that living corals are vulnerable to synergistic effects of past fishing pressure and human population density. The delayed effect of fishing pressure suggests

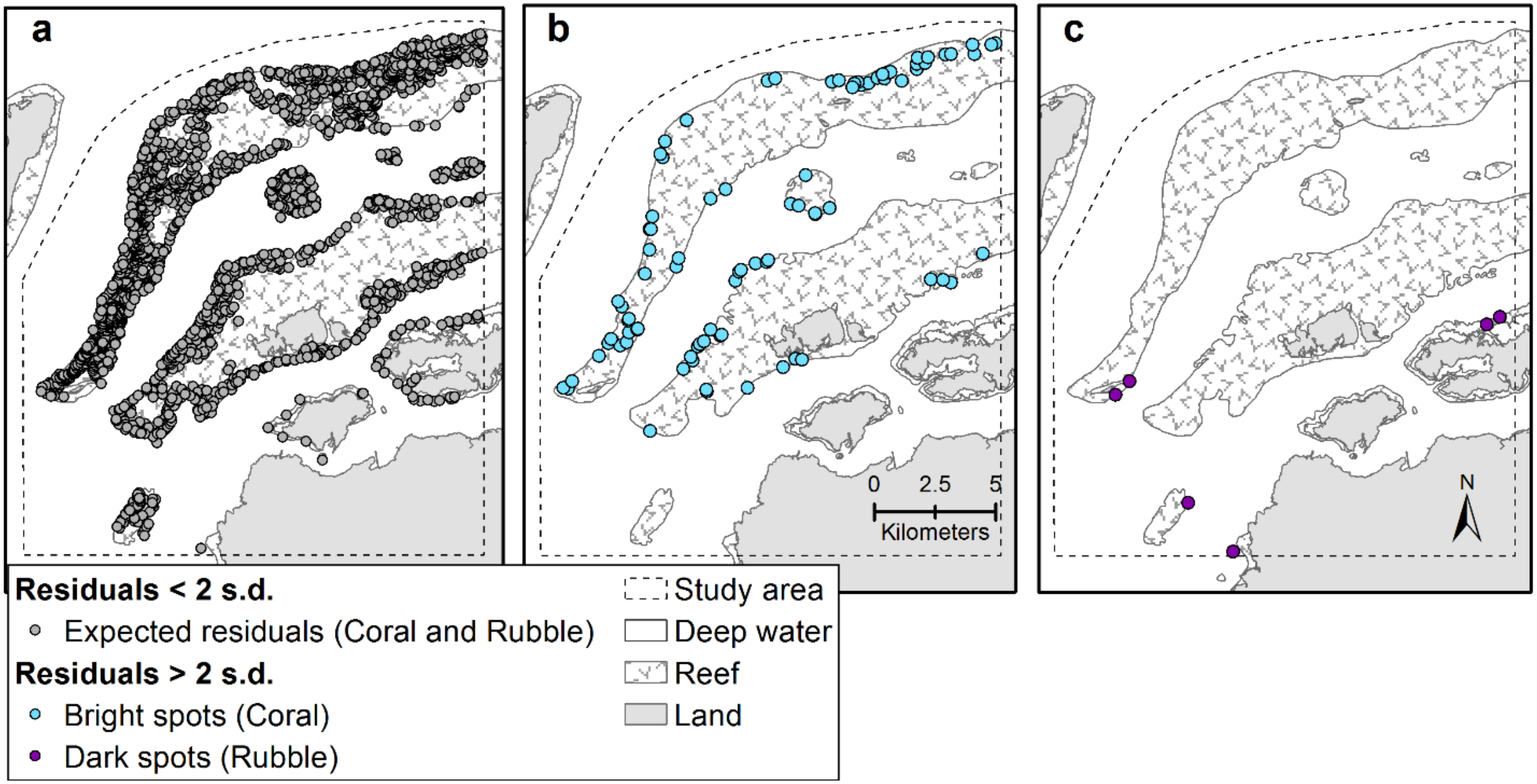

Figure 5.5 Maps highlighting locations in Danajon Bank, Philippines where the presence of living coral and rubble (a) fell within an expected range (< 2 standard deviations above or below expected values; grey locations); and (b-c) deviated from the expected reef state > 2 standard deviations (blue and purple locations). Bright spots (blue) were dominated by living coral, but were predicted to be rubble-dominated. Dark spots (purple) were dominated by rubble, but were predicted to be coral-dominated. Maps correspond to the model in Figure 5.

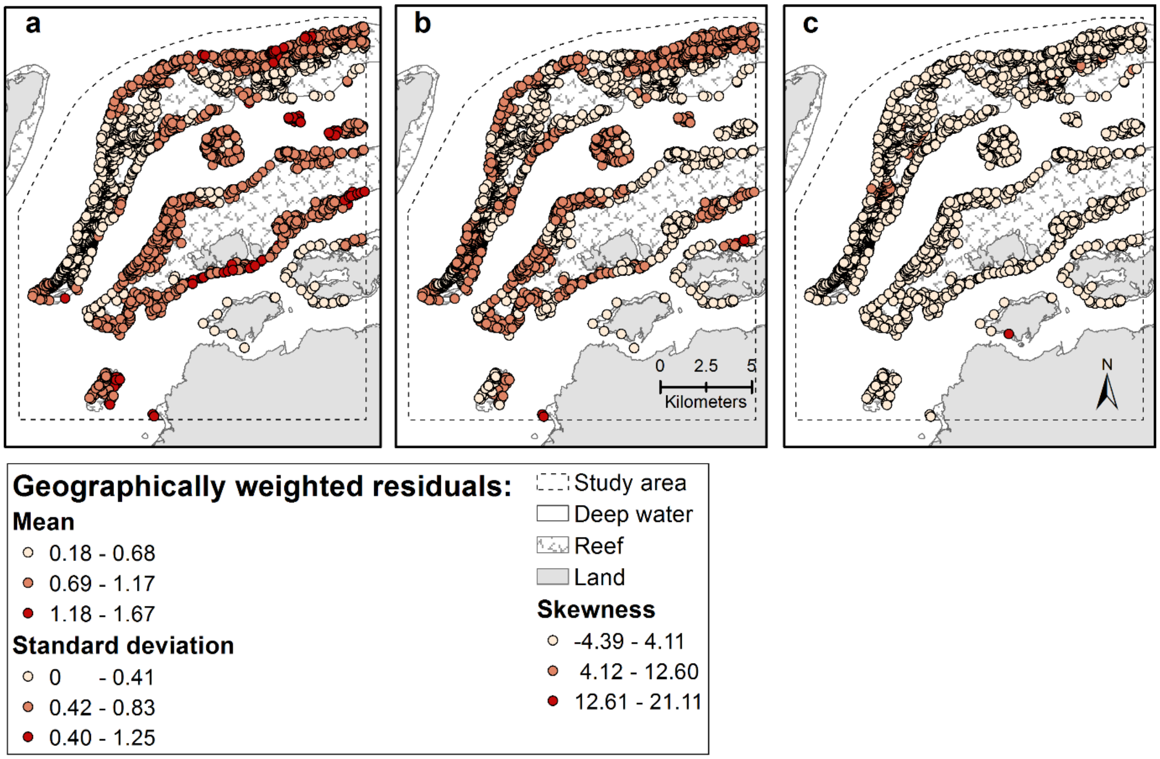

Figure 5.6 Geographically weighted statistics of residuals from a model evaluating the probability of living coral being present in the Danajon Bank, Philippines. Maps correspond to the model in Figure 5.4. Residuals > 2 are considered outliers. (a) Geographically weighted mean; (b) Geographically weighted standard deviation; (c) Geographically weighted skewness.

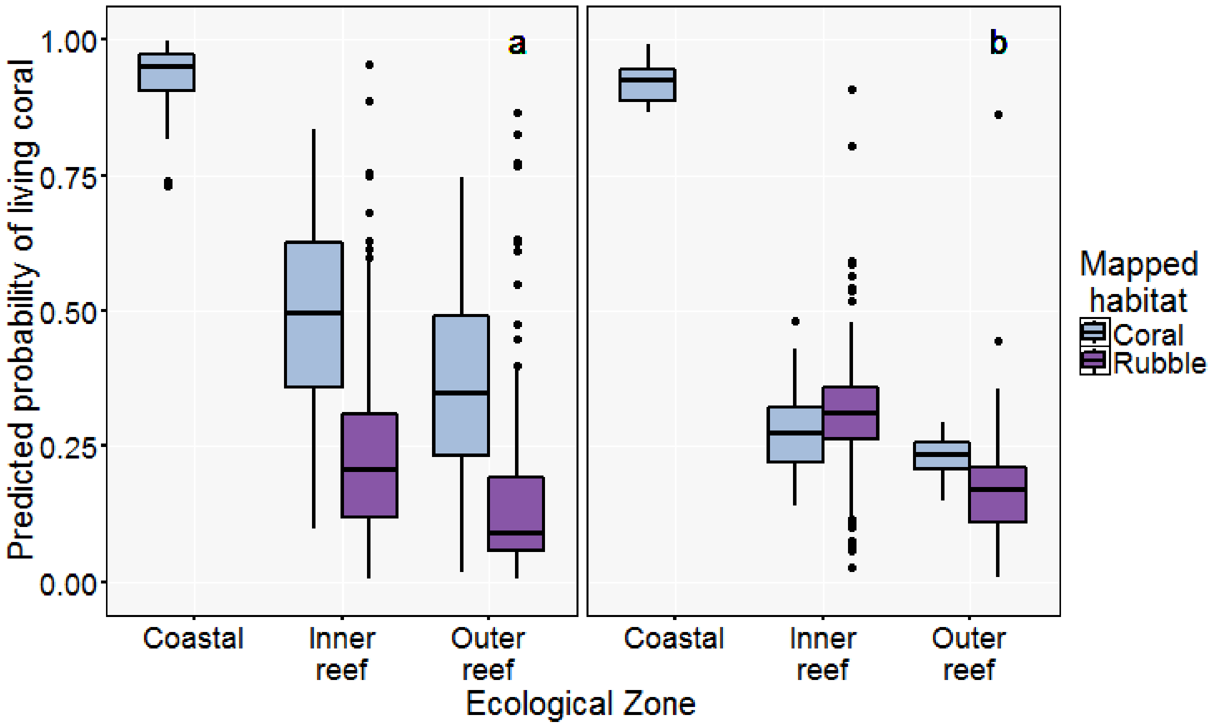

Figure 5.7 Differences in the ability of two models to distinguish locations dominated by living coral from those dominated by rubble. Models tested using independent samples from habitat map. (a) Final model included anthropogenic drivers, depth, and landscape variables. This figure corresponds to the model in Figure 5.4a. (b) Simplified model included anthropogenic drivers and depth, but excluded landscape variables.

that corals responded slowly to this widespread stressor. Additionally, reefs were less likely to harbor living corals in areas with high levels of blast fishing. In contrast to the negative effects of fishing and human population density, I found that living corals were more likely to be present in MPAs and in deeper waters. The small, but positive influence of protected areas supports previous evidence that MPAs benefit corals. This finding was particularly interesting in the Danajon Bank because several MPAs were established in highly degraded locations (A. Vincent, pers. comm). Furthermore, our work demonstrates that maps created from high spatial resolution satellite images can be used to evaluate the effects of MPAs on corals. Overall, I found the strongest relationship between coral presence and the spatial arrangement of habitats. Landscape pattern indices – particularly compact habitat patches – helped identify areas most likely to support living coral. Thus, areas with compact habitat patches in deeper waters appear to have greater potential to function under stress, particularity if they are located in protected areas. These relationships offer guidance for management of ecosystems in the face of growing pressures, which I discuss below.

5.5.1 Interacting and long-term effects of multiple stressors

One large gap in cumulative stressor research involves interactions between fishing and other disturbances (e.g. nutrient pollution, climate change), which is of high conservation significance given the widespread footprint of fishing and the documented impacts of fishing on marine ecosystems (Worm et al. 2006, Swartz et al. 2010). I found that the influence of fishing pressure from all small-scale fishing gears was greater in areas with higher human population density. Unpacking the individual effects of human population density would reveal individual threats such as trampling, gleaning by women fishers, coral harvesting, and untreated sewage (nutrient pollution). While the mechanism for this synergy remains unknown, there are many potential reasons for a decrease in coral cover including the ability of algae to outcompete corals when nutrient pollution is coupled with the overfishing of herbivores (Jompa and McCook 2002), and the reduction in structural complexity from chronic impacts to reefs (e.g. fishing, tramping) which in turn reduces reef fish density and biomass (Hawkins and Roberts 2004, Graham and

Nash 2013). In the Danajon Bank, where blast fishing persists despite a century-old effort to eliminate it (Alcala and Gomez 1987), I found that this destructive method continues to reduce the probability that an area supports living corals. However, in locations where blast fishing was practiced near highly population areas, blast fishing and population density exhibited an antagonistic effect. This antagonistic relationship may exist because corals killed by blast fishing could not be further damaged by other stressors (Côté et al. 2016) or may be a spurious effect.

The relationships I documented among long-term fishing pressure and coral distributions suggests heavy fishing pressure affected reefs indirectly, while destructive fishing affected reefs directly. Although lag-effects are widely recognized in complex adapted systems, they can be challenging to document and easy to overlook (Walker and Salt 2006). I found that fishing pressure from as far back as 1980 influenced the probability that corals were present at a location in 2010. However, the effects of contemporary fishing were not informative of coral distributions. I believe this pattern represents a time-lag response of corals to fishing pressure. Overfishing in the Danajon Bank and elsewhere has reduced populations of herbivores including parrotfishes (Scaridae), rabbitfishes (Siganidae), and sea urchins (e.g. *Diademia spp.*) (Green et al. 2000, McManus and Polsenberg 2004). Such herbivores are important to reef resilience because they promote coral recovery and control algae (Mumby et al. 2007, Olds et al. 2012c). Heavy fishing of herbivores can reduce herbivory and cause corresponding declines in living corals and coral recruitment (McManus and Polsenberg 2004, Mumby et al. 2006, Nyström et al. 2012). The effects of overfishing herbivores are widely documented in fine-scale experiments and diving surveys (e.g. Mumby et al. 2007, Lokrantz et al. 2010, Olds et al. 2012a), but to our knowledge this is the first time the indirect effects of fishing have been documented at the scale of a reef-system. The accumulation of fishing impacts over time was also evident for blast fishing. I found that contemporary blast fishing had a negative influence on corals, but effects were stronger if I considered the cumulative pressure of blast fishing from 2000 and 2010. Blast fishing kills corals directly, so it follows that the direct effects of blast fishing would be immediately visible using the 2 m resolution of WorldView2 satellite images.

5.5.2 Corals more abundant in protected areas

I found a small, but significant positive relationship between MPAs and the probability that an area supported living coral. MPAs can benefit corals through several mechanisms, including by restoring communities of reef-dependent organisms (Bellwood et al. 2004) and by increasing grazing by herbivores (Mumby et al. 2007). Over the past several decades, coral cover has declined globally. MPAs have been able to slow coral declines and ultimately stabilize coral cover within their boundaries (Selig and Bruno 2010). However, protected areas are still vulnerable to regional and global stressors (Graham et al. 2007, McMenamin et al. 2008), including sedimentation, disease outbreaks, and ocean warming. Our work indicates that protection corresponded to a higher probability that an area supported living corals. A time series of coral habitats will be necessary to determine if MPAs in the Danajon Bank are promoting coral recovery, maintaining existing coral cover, or slowing rates of coral loss.

5.5.3 Depth and landscape features increase the probability of living coral

Our research supports growing evidence that increasing depth has a positive influence on corals (Berkström et al. 2013, Graham et al. 2015). I found increasing depth from 0-15 m was an important predictor of living corals. To date, coral reefs have been documented to have higher abundances of reef fishes with increasing depth (e.g. from 0-10 m (Berkström et al. 2013); from 10-70 m (Lindfield et al. 2016)). Moreover, there is evidence that reefs possess a higher resilience to climate-driven coral bleaching at depths greater than 6.6 m (Graham et al. 2015). Despite the accumulating evidence that increasing depth benefits coral reef systems, the mechanisms behind this pattern remain unclear. Depth may correlate with other beneficial factors such as reduced light stress, lower temperatures, lower levels of physical human impacts, or lower algal growth (McCook et al. 2001, McClanahan et al. 2012). During interviews, some fishers suggested that some of these factors may be at play in the Danajon Bank. For example, many respondents reported that the outer reef zone was seasonally exposed at low tide and that one fisher – who I did not interview as he had recently passed away – used that tidal pattern to fish by walking and pouring buckets of cyanide on the reef. Such patterns would create additional stressors in the outer reef.

By integrating landscape variables into this research, I take a distinctive and informative approach to understanding the influence of marine stressors on the ocean. Our understanding of seascape ecology – the application of landscape ecology's frameworks and concepts to submerged ecosystems – remains a poorly understood aspect of marine and coastal ecology (Pittman et al. 2011). To date only a handful of studies in coral reefs have examined the influence of landscape pattern on ecological dynamics, with most research focused on the influence of landscape patterns on reef fishes (e.g. Dorenbosch et al. 2007, Grober-Dunsmore et al. 2007, Berkström et al. 2012). I found that landscape features were essential for accurately predicting the distribution of living corals. Indeed, when models included depth and anthropogenic activities alone (e.g. fishing, population pressure, and MPAs), they had no predictive ability. This work contributes to emerging evidence that landscape dynamics influence corals. In one example, the spatial arrangement of Australian reef habitats affected fish feeding patterns, thereby facilitating trophic cascades that increased coral settlement (Olds et al. 2012c). However, corals may not always respond the same as fishes. For example, connectivity to seagrass can benefit coral reef fishes (Olds et al. 2012b, Berkström et al. 2013). In contrast, I found that the probability that an area supported living corals increased with distance from seagrass. This supports general ecological zonation patterns on reefs whereby corals and seagrasses benefit from different conditions (e.g. corals prefer high water clarity while seagrasses occupy shallow, sediment-dominant systems).

5.5.4 Addressing uncertainties in variables

Accurately modeling ecological relationships is influenced by available data. The variables included here ranged in precision and spatial resolution, as is typical of variables collected at local vs. global scales. This thesis focused on the relationship between habitats, fishing, and demographic trends and I collected these variables at a high spatial resolution (e.g. 2 – 10 m). There are however, limitations to any data sources. For example, data for fishing effort and village populations have characteristic limitations of LEK data (e.g. LEK is more accurate for recent than past data; see further discussion in Chapter 4). In contrast, global data have lower spatial resolution and may not accurately capture local patterns. For example, data for aragonite saturation and past thermal stress data were at a coarse spatial scale and did not vary across the study area. These data were modelled at 200 km and 1 km resolution respectively (Table 1). Other spatial variables that may affect corals (e.g. *chl a*; temperature) did not exist for the Danajon Bank. Higher spatial resolution data will be necessary to understand how these stressors interact with fishing and other pressures.

5.5.5 Implications for coral reef monitoring and conservation

Monitoring and evaluation are essential for understanding and offsetting the effects of global change on corals (Darling et al. 2010), and other critical species. In the last decade, estimated yearly coral cover loss in the Indo-Pacific was 3,168 km^2 per year (Bruno and Selig 2007), and 2016 is anticipated to have record coral losses due to the longest global coral bleaching event on record (http://www.globalcoralbleaching.org/). Despite the benefits of understanding ecological responses to stressors, it has been an ongoing challenge to developing research and monitoring techniques that provide insights at appropriate spatial and temporal scales. This difficulty is particularity evident for local stressors. Efforts to monitor corals– or other marine habitats – over large areas can benefit from the recent explosion of high resolution satellites that can identify corals, including in places which are not directly surveyed (Roelfsema et al. 2013, Selgrath et al. 2016). In situations where ecosystem-scale monitoring does not exist, another viable option for evaluating stressors is drawing upon alternative data sources, such as the LEK used here (Reed 2008, Pauly and Zeller 2016). LEK can provide insight into past stressors whose presence was not formally documented and its value has been met with increasing recognition by the scientific community (Thornton and Scheer 2012). The growing number of examples where western science is integrated with LEK illuminate LEK's potential for robust insight generation. For example, I demonstrate, to our knowledge, the first example of satellite image based maps and LEK being integrated to detect the influence of MPAs and fishing at the scale of ecosystems.

Characterizing areas that have withstood or adapted to existing stressors, as I have done here, can reveal windows of opportunity for conservation. The strong relationship I documented between the presence of living corals and seascape conditions demonstrates the importance of approaching ecosystem conservation with a landscape perspective. Our findings have five management implications for coupled human-natural systems. First, beneficial landscape characteristics can be incorporated into marine spatial planning. For example, new protected areas in reefs could prioritize locations with compact coral patches. Second, identifying bright spots provides an opportunity to characterize locations within an ecosystem that fare better than expected under stress and these characteristics can inform management planning. Third, locations with overlapping and synergistic stressors provide opportunities for high conservation gains. In these locations, reducing one of the two synergistic stressors may lead to larger benefits than reducing the stressors where they occur independently. For example, our findings suggest that reducing fishing in locations near high human populations may provide greater benefits to corals than the same reductions would provide in isolated areas. This approach, however, may be challenging given the high dependence of communities on fishing in nearby waters. Taking another strategy, sites that have relatively low fishing pressure and are located far from population centers would be alternative candidates for protection. Fourth, our findings reiterate the importance of phasing out highly destructive fishing methods, such as blast fishing. Finally, when ecosystems respond slowly to stressors – as I show corals did to fishing pressure – it will be critical for monitoring to account for delayed effects of stressors, which may continue to develop after protections have been put in place. Community outreach must ensure resource users make informed management decisions regarding stressors that have slow moving, but significant effects on ecosystems (Walker and Salt 2006, Ostrom 2009).

In this study, I have demonstrated the effectiveness of integrating LEK and satellite mapping for monitoring the relationship between stressors and coral at the scale of ecosystems. I show that patterns previously observed at single, local sites (e.g. the positive influence of depth on reef communities) scaled up to systems and that landscape or seascape patterns in patch compactness influenced the probability of corals being present at local sites. Moreover, I found that long term fishing pressure has a delayed, but significant influence on corals. Going forward, this approach can be used to monitor how reefs are changing under the rise and fall of interacting stressors. As we continue to build our understanding of stressors and their role in shaping ecosystems, new technologies and local knowledge provide opportunities to assess the effects of long-term stressors and capacity of ecosystems to adapt to environmental change.

Chapter 6: Conclusion

6.1 Introduction

My research aimed to fill three large gaps in cumulative stressors research: the long-term influence of fishing, the ecological effects of interactions between fishing and other disturbances (e.g. nutrient pollution, climate change) (for example, see references in Ban et al. 2014), and the impact of landscape patterns on these interactions. First, developing a more comprehensive understanding of fishing and other stressor interactions is of high conservation significance given the widespread footprint of fishing and the documented impacts of fishing on marine ecosystems (Pauly et al. 1998, Swartz et al. 2010). Second, cumulative impact mapping has currently focused on documenting overlapping impacts, with a goal of moving into a more direct assessment of how co-occurring impacts in turn influence ecological conditions (Ban et al. 2010, Halpern et al. 2015). Third, understanding seascape patterns, such as patch compactness, remain an exciting research frontier because there is a broad lack of knowledge about the influence of seascape patterns on marine and coastal ecology (Pittman et al. 2011).

My research offers a rare in-depth look at the spatial and temporal development of small-scale fishing and uses this information to evaluate the impacts of fishing and other stressors on coral reef ecosystems (Figure 6.1). Small-scale fishing is one of the major influences on coral reef ecosystems (Johnson et al. 2013). I used the Danajon Bank coral reef ecosystem in the Philippines as a case study to evaluate how spatial and temporal trends in small-scale fisheries had affected ecological conditions. Degradation in the Danajon Bank has been widely documented (Green et al. 2000, Marcus et al. 2007), but the long-term changes in fishing gears, total fishing effort, and spatial fishing patterns which have contributed to degraded conditions have never been quantified. I found that fishers of Danajon Bank greatly diversified their fishing gears (1950-2010) (Chapter 2). In many ways individual fishing practices remained steady during this period (e.g. the number of fishing gears used by individuals annually and the mean individual fisher effort demonstrated only slight changes).

The effort distribution among fishing gears has changed, however. Through the late 1990s, there were increases in the use of intensive gears, particularly non-selective, active, and destructive fishing gears (Chapter 2). Over the same period, fishing effort using hook and line gears and traps was relatively steady, while effort with nets and diving greatly increased (Chapter 2). In contrast, blast fishing effort decreased in the 1960s and then rose slightly thereafter. The implementation of new fishing regulations in 1998 reduced growth in the proportion of fishers using these gears, but had a more negligible impact on the total effort with intensive gears. My analysis of small-scale fishing effort in the Danajon Bank quantified, to my knowledge for the first time the magnitude of the increase in fishing intensity over the past 50 years (1960-2010). There are numerous reports documenting declining CPUE after the 1970s or 1980s, but the total levels of fishing effort remained unknown (Chapter 3) (e.g. Sotto et al. 2001, Green et al. 2004, FISH Project and USAID 2005). Unpacking this change revealed that fishing intensified in five ways: (i) a rapid increase in cumulative number of fishing days, driven by significant growth in the number of fishers; (ii) an expansion of fished areas; (iii) a diversification of fishing gears; (iv) a significant proliferation of intensive fishing gears; and (v) an increase in the spatial extent and overlap of gears (Chapters 2 and 3).

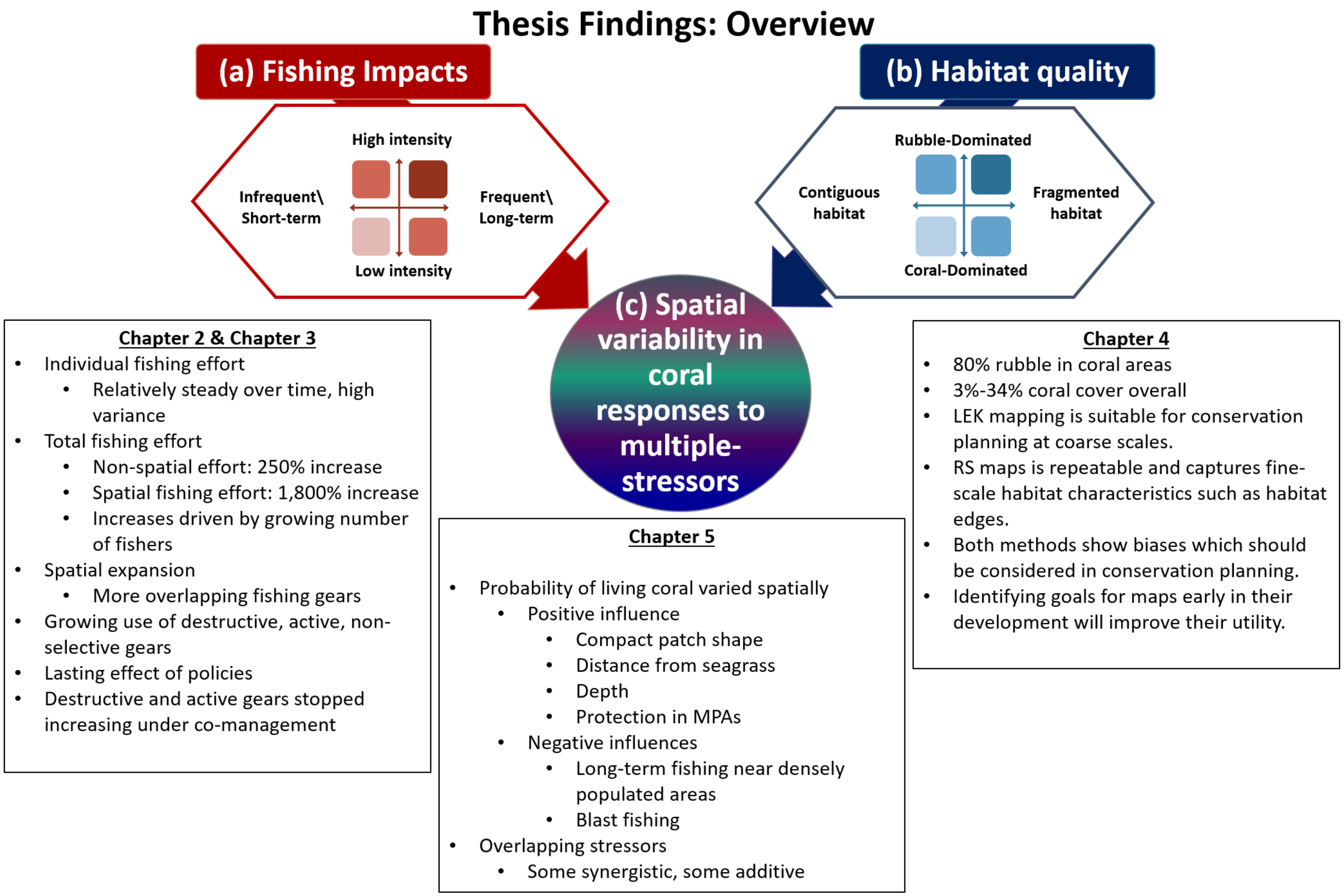

Figure 6.1 Schematic outline of thesis structure, with main findings from data chapters: (a) Chapter 2 and Chapter 3; (b) Chapter 4; and (c) Chapter 5.

Turning to look at the ecosystems these fisheries impacted, I found that local environmental knowledge (LEK) and remote sensing (RS) are both useful for conservation planning, yet the ideal approach depends on the goals for the map. Maps derived from LEK are most suitable for conservation planning at coarse scales, while RS captures fine spatial scale habitat characteristics such as habitat edges (Chapter 4) and is more easily repeatable at a higher frequency. My novel integration of satellite imagery and participatory mapping was critical in evaluating the spatial dynamics and long-term influence of stressors on vulnerable coral reef ecosystems (Chapter 5). Through this research, I found that the probability that living corals were present at a location was strongly correlated with the spatial arrangement of habitats. Additionally, I found that living corals benefited from increasing depth and protection in reserves, while long-term fishing pressure impacted corals negatively.

For each chapter in turn, I now comment on (i) my research question and methods and (ii) my scholarly contributions. I then discuss limitations of my approaches. I conclude by describing practical applications of my research, broader conservation implications, and directions for future research.

6.1.1 Do historic trends in fishing gear use suggest intensification, and how were these changes correlated with fisheries governance? (Chapter 2)

My work in Chapter 2 provided a novel long-term integration of fishing and policy changes and their impact on the diversity and intensity of small-scale fishing gears. Over the past 30 years, the majority of research on small-scale fisheries in coral reefs has focused on direct effects to fish (e.g., diversity of fish communities or catches and/or fish population density (Johnson et al. 2013). Although the importance of evidence-based fisheries management is widely touted, only a fraction of small-scale fishing research critically evaluates long-term changes in the fisheries as a social-ecological system. Much research ignores the changing patterns of gear use and the influence of management and governance (Johnson et al. 2013). Variations in the types of gear (as well as effort) must be managed directly because gears influence which species are caught and the collateral impacts to ecosystems (McClanahan and Cinner 2008). There is evidence that fisheries in Europe have evolved by shifting to use more avaricious, damaging gears (Thurstan and Roberts 2010). For small-scale fisheries, however, the evolution of fishing gear-use in response to ecological declines remains undocumented at both the individual fisher and system level. Thus, my research in Chapter 2 represents the first attempt to document how the suite of fishing methods in a diverse, coral reef system have adapted to previously documented CPUE declines (Pauly and Chua 1988, Sotto et al. 2001, Green et al. 2004).

Not only do my findings expand our knowledge of how the use of fishing gears has evolved over time, but they also provide insights into gear-based management – a major knowledge gap for small-scale fisheries (Johnson et al. 2013). At the level of individual fishers, I found that fishers rarely abandoned familiar gears (1950-2010). This strong inertia of familiar methods could have both negative and positive ramifications. The stickiness of familiar fishing gears may challenge programs that aim to help fishers switch to sustainable methods. However, the stickiness also creates a high possibility that changes in gears will persist once they become established. Although fisheries governance in the Philippines has aimed to reduce non-sustainable fishing practices for nearly two decades, my research found that much more work is needed to meet this goal.

At the whole system level, I found that the impacts of fishing intensified over time, due to gear diversification and growing effort by destructive, active, and non-selective gears. The proportion of fishers using intensive gears slowed after 1998, when many intensive fishing gears became illegal and community-based management was established. That said, total effort of intensive and illegal fishing gears is still high, and increasing. There has been a growing emphasis on using co-management as a tool to improve the sustainability of fishing practices (Pinkerton 1994, Gelcich et al. 2010, Evans et al. 2011, Wamukota et al. 2012). While co-management can be effective in many cases (Berkes 2009), my findings add more evidence to concerns that these laws are widely ignored in many communities in the Philippines (Muallil et al. 2014). I identify one possibility for improving the uptake of these laws is improved participation in fisheries management groups. Such participation, I found, is currently active in some communities yet absent in others.

6.1.2 How have the spatial dimensions of fishing effort changed? (Chapter 3)

In Chapter 3, I compared metrics of fishing effort using non-spatial vs. spatial approaches and explored the spatial dynamics of small-scale fisheries over five decades. Heavy fishing pressure in the Philippines, and many other developing countries, has proven nearly impossible to reduce and therefore remains an urgent challenge. However, I have a limited understanding of how fishing activities have changed over time. The evolution of fishing practices has been described qualitatively or estimated, but rarely measured (e.g. Anticamara et al. 2011, Johnson et al. 2013). Participatory mapping by fishers is an established approach that has the potential to benefit fisheries management (Thomas-Slayter 1995, Hind 2015). I took the unusual step of integrating well-established participatory mapping techniques (low technology) with 10 m x 10 m spatial resolution satellite imagery (high technology). I arrived at this decision after a pilot study, using nautical maps, generated maps that were less precise than I wanted. Furthermore, research at that time suggests satellite imagery had the potential to improve the accuracy of participatory mapping of fisheries (Hall and Close 2007, Moreno-Báez et al. 2010).

Decreasing intensive fishing practices, effort, and area impacted are all necessary components of improving the sustainability of small-scale fisheries in the Philippines and elsewhere. I demonstrated that from 1960-2010 spatial estimates of fishing effort (total days per year fished) exhibited a rapid, yet variable increase that was 7.2 to 8.6 times greater than estimates provided by non-spatial data alone. My research revealed that contemporary small-scale fishing pressure in 2010 was 1,800% higher than fishing levels in 1960. This large increase in fishing effort was entirely driven by the rapid increase in the number of fishers. Moreover, heavy fishing pressure affects significantly more of the ocean today than in the past. This paints an urgent picture that parallels estimations of overfishing in other small-scale fisheries, ranging from the Caribbean (Hawkins and Roberts 2004) to East Africa (Lokrantz et al. 2010).

6.1.3 How can we map coral reef habitats quickly and effectively? (Chapter 4)

In Chapter 4, I compare the value of local environmental knowledge (LEK) and remote sensing (RS) approaches for mapping coral reef habitats. Habitat maps provide a foundation for many conservation programs and can be used as effective surrogates for species conservation (Margules and Pressey 2000). Despite their utility, these maps are rarely available for marine systems due to the technological difficulties of mapping underwater habitats. Furthermore, technological limitations have restricted the conservation utility of existing satellite image-based maps. For example, maps have focused on small areas (e.g. single reefs) and/or the maps' spatial resolutions were too low capture ecologically-relevant variations in habitats (e.g. Landsat at 30 m resolution) (Palandro et al. 2003, Scopélitis et al. 2009, Hamel and Andréfouët 2010). The WorldView-2 satellite was launched at the end of 2009, during my PhD. This advanced satellite captures high spatial resolution imagery (pixel sizes 2 m – 10 m) with spectral bands more suitable for marine applications (blue and/or green wavelength ranges, e.g. WorldView-2) (Roelfsema et al. 2013b). To date, however, high spatial resolution mapping of coral systems exists in only a handful of areas (Hamel and Andréfouët 2010). I integrated this new technology into my research to explore the capacity of this new method to create benthic community maps of coral reefs at finer spatial scales (2- 5 m) over large areas (> 300 km^2).

When considering the accuracy, cost, and conservation utility of LEK and RS, I found that both approaches were suitable for mapping coral reef habitats, yet each exhibited various strengths and weaknesses. Both LEK and RS methods had accuracies that were above the standard for marine remote sensing maps (i.e. > 60%, Roelfsema and Phinn 2013). However, the RS-derived map had higher overall accuracy (LEK: 66%, RS: 76%). Interestingly, the two approaches produced maps, which were largely dominated by different habitats (LEK: rubble, RS: seagrass). This difference is likely due to methodological biases (Foale 1998, Lauer and Aswani 2008, Roelfsema and Phinn 2008): LEK tends to focus on aspects of the ecosystem that are locally important for fishing or other uses, and RS has limitations when mapping habitats in water with low levels of visibility. When considering costs and benefits for conservation programs, the LEK method was less expensive, less technical, and had the benefit of engaging local community members. The RS method provided finer detail of habitat arrangements (e.g. identifying habitat edges; documenting small habitat patches), has potential for regular repeat monitoring (without the risk of interview fatigue), and can monitor areas unknown to fishers. This research represents the second example – globally – of mapping corals at an ecosystem-scale using object-based image analysis (OBIA) in conjunction with high resolution satellite images (Roelfsema et al. 2013, Selgrath et al. 2016). Furthermore, this methodological breakthrough offered the first instance of a direct comparison of the ecological and financial tradeoffs between LEK and RS mapping of coral reef habitats. In doing so, my research provides practical guidance for conservation-oriented habitat mapping (Margules and Pressey 2000).

6.1.4 Can we use the spatial distribution of anthropogenic stressors and biophysical conditions to predict the presence of living corals? (Chapter 5)

Seascape ecology can be an informative approach for evaluating the fate of stressed coral ecosystems. To date, however, seascape variables are rarely included in evaluations of coral reef resilience. Despite the heterogeneity of the Danajon Bank, I was able to model the distribution of corals based on biophysical conditions and anthropogenic stressors. I found that the probability that an area supported living corals increased with greater: compactness of habitat patches; isolation from seagrass; depth; and with protection in marine reserves. The strong landscape-coral relationship I demonstrated – and the significant influence of seascape patterns reported elsewhere – from Australia (Olds et al. 2012c) to East Africa (Berkström et al. 2013) to the Caribbean (Grober-Dunsmore et al. 2009) – provide growing evidence that seascape ecology approach is quite informative. Elsewhere there has been an increasing recognition that increasing depth correlates with positive attributes of coral reefs. For example, Graham et al. (2015) reported that depth and initial structural complexity were consistent predictors of coral or macro-algal dominated reefs post disturbance in East Africa and the South Pacific. This is the first study to my knowledge, evaluating ecological effects of multiple stressors on coral reefs with direct quantification of fishing pressure (see references in Ban et al. 2014). Moreover, I found that the relationship between total fishing pressure and the probability of an area supporting living corals showed a time-lag. This time-lag I documented corroborates evidence from fine-scale surveys and experiments that fishing can influence reefs indirectly (e.g. reduced herbivory, reduced coral growth, predation) (Graham et al. 2007, Lokrantz et al. 2010, Darling et al. 2010). This demonstrates that high resolution remote sensing is a viable tool for directly evaluating the short-term and long-term impacts of highly destructive fishing gears on coral reefs.

6.2 Limitations

My findings were undoubtedly influenced by limitations relating to bias in LEK data gathering and satellite technology. Participatory mapping can be influenced by recall bias, especially in historical assessments (Means and Loftus 1991) as well as the under-reporting of illegal practices (Gavin et al. 2010). As events become more distant in a respondent's past, there is more chance of error. I addressed this bias through methods such as personal timelines and a focus on fishing activities (gear use and effort) which changed gradually over time and were not influenced by windfall events (e.g. unusually large catches) (Neis et al. 1999). Without reference data, it is not possible to evaluate the strength of this bias (O'Donnell et al. 2012). I proceeded in the spirit that it was better to be approximately right by using LEK data than to be precisely wrong by not making any effort to document past trends (Chambers 1994, Johannes 1998).

A second type of LEK bias is the tendency for respondents to under-report illegal practices. During a pilot study, I determined that respondents were far more likely to discuss illicit practices when they felt a sense of connection to the people conducting the interviews. I adapted my research protocol to accommodate this observation in two ways: I invested a significant amount of time developing rapport with the respondents during the initial portion of the interview and I hired research assistants from the region. Further, I conducted interviews with individual fishers in an attempt collect unbiased data about past fishing (Regan et al. 2006).

The image analysis approaches I incorporated into my research evolved in response to technical limitations and emerging technology. My original intent, using a time series of SPOT-5 satellite images to map coral over time, was found to be lacking as the spatial and spectral resolution was poorly suited for coral reefs (< 50% accuracy). The Worldview-2 satellite, launched mid-way through my dissertation, with a higher spatial resolution and a more optimal spectral resolution, greatly improved the quality of the habitat maps. However, satellite mapping has some inherent limitations. One limitation is that reduced light penetration limits RS to shallow and/or clear waters for many sensors (Mumby et al. 1998). Another limitation is that I was able to conduct field calibration of the map using a combination of point intercept transects (PIT) and spot-check surveys using bucket-viewing. These two ground verification methods provided a practical balance of allowing us to evaluate some areas in very high detail, yet still assess the broader area with less detail. The accuracy of the remote sensing map, however, might have been higher if only the PIT method was used because it provides a higher level of spatial resolution than spot check surveys (Roelfsema and Phinn 2008). Additionally, the more recent ease and availability of obtaining georeferenced photographs during PIT, could further improve ground verification procedures. Finally, I conducted a ground verification using independent data from manta tow surveys, but had access to such data on only small portion of the map.

6.3 Conservation implications

From my spatial and temporal documentation of extractive pressures, it is clear that small-scale fisheries are not the universally sustainable alternative to industrial fisheries that they are frequently characterized to be (Jacquet and Pauly 2008). The widespread, pressing influence of fishing on Danajon Bank provides a strong reminder that activities that may have been “sustainable” when practiced by a few people can become incredibly damaging when practiced at higher frequencies. Moreover, the increasing use of diverse and damaging fishing gears that I documented paints a clear picture that small-scale fishing methods have intensified in spite of and because of the rapid increase in fishing effort. This is not to say that small-scale fishers are inherently nefarious or inextricably trapped by their circumstance (Béné et al. 2010). However, there are significant challenges to overcome if I am to improve the sustainability of these critically important fisheries.

Given the high levels of fishing pressure I documented on Danajon Bank, it is impressive that a fishery is still viable and that biodiversity remains notable – though declining (Nañola et al. 2011) – in the region. I estimate that over 11,000 men currently fish (full- and part-time) in the 19 km x 22 km area. This estimate does not include women, who comprise 42% of all fishers and catch about one quarter of the catch mass extracted from Danajon Bank (Kleiber et al. 2014). Using these numbers, I estimate nearly 15,000 adults currently fish in my study area alone. For comparison, the Canadian federation of independent operators has 18,000 members and it is estimated that around 30,000 people work in Canadian small-scale fisheries.

The high levels of fishing carry ecological and social consequences. Catches have declined (Sotto et al. 2001). Intensive fishing has led to biodiversity losses (Nañola et al. 2011) and habitat degradation – as documented in my thesis and elsewhere (Chapters 3, 5; e.g. Gomez et al. 1994, Roberts et al. 2002, Marcus et al. 2007). I found that 80% of corals had turned into rubble or were covered with algae. In response to this sweeping degradation, fishers have found ways to wring anything that remains out of the ocean. Indeed, “*makadaginot*” the Cebuano word describing the ability to squeeze out a little more from something is a word that 25 respondents used to describe their most important fishing grounds. The desperation of fishers is manifested in their high levels of poverty (PPDO Bohol 2013) and their willingness to take on high risks (Selgrath et al. 2014).

Next, I discuss how my research can be used to support broader conservation goals using seven mutually supporting steps (Figure 6.2). Here I use the small-scale fisheries I studied as an example, but these steps can be adapted for reigning in the spiraling intensification of other stressors.

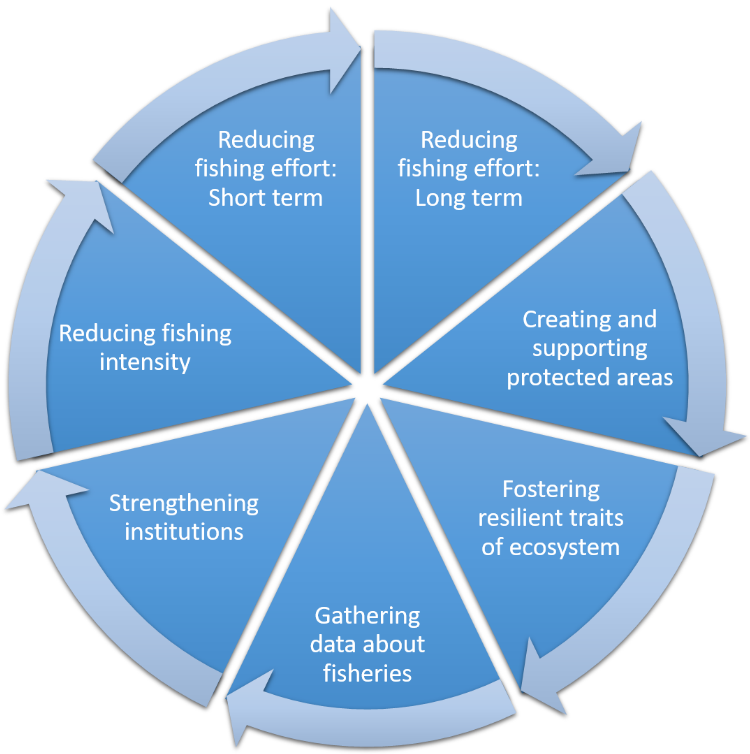

Figure 6.2 Improving conservation success by reducing the effects of stressors.

6.4 Recommendations

Below I discuss seven recommendations for improving conservation success (Table 6.1; Figure 6.2). With these recommendations, I recognize that in the context of many small-scale fisheries, problems of overfishing are tightly bound to larger structural issues including: poverty; limited access to health, education, and other services; ineffective governments; limited opportunities; and social marginalization (FAO 2015). Thus I present recommendations here, while recognizing that fishers are situated within a larger, multi-dimensional system.

Table 6.1 Seven recommendations for improving conservation success by reducing social-ecological stressors.

Step	Goal	Supporting Actions
Reducing fishing effort: Short-term		
	Reduce local fisher populations, short term	
		Develop programs to support fishers moving to cities
		Improve reproductive health to reduce accidental pregnancies
		Job training
		Support networks for safe transition to cities
	Reduce individual fishing effort	
		Financial incentives
		Alternative livelihoods
	Limiting access to fishery	
		Ending reliance on the ocean as employment insurance for the jobless
		Defining who is authorized to fish in the Danajon Bank
		Documenting all full- and part-time fishers through a universal fisher registry
		Strengthening village level institutions such as fishers' organizations that provide accountability around who fishes and how they fish
Reducing fishing effort: Long-term		
	Slow population growth	
		Improve reproductive health to reduce accidental pregnancies
		Reduce teenage pregnancies
Reducing fishing intensity		
	Fully implementing dormant laws supporting conservation	
		Eliminating the use of damaging gears
	Limiting entry to the fishers	
	Developing greater accountability and self-monitoring	
		Limiting entry to the fishery and eliminating the practice of short-term fishing migrations
	Improving communication amongst fishers so that they hold each other accountable	
	Strengthening enforcement	
		Improving officer training
		Providing long-term funding

Step	Goal	Supporting Actions
		Engaging community members to report illegal activity
		Supportive government leaders
Strengthening institutions		
	Fishers' organizations	
		Improve fisher knowledge of, and compliance with, existing legislation
		Create social pressure to fish sustainability
	Strength of municipal-level governance	
		Provide coordinated support and education to village-level fishers' organizations
		Formalized and professionalize municipal-level fishery positions
	Regional initiatives	
		Create institutions that cross legal boundaries to holistically consider ecosystems
	National and provincial initiatives	
		Support village level organizations with extension support and other resources
	Outside partnerships	
		Religions institutions
Creating and supporting protected areas		
	Existing MPAs	
		Strengthen enforcement and compliance
		Engage communities
	Expand MPAs	
		Use Fisheries Code protection requirements as a resource
		Engage communities
Fostering resilient traits of ecosystems		
	Enhancing the capacity of ecosystems to absorb disturbances	
		Identify important biophysical traits
		Include important biophysical traits in marine spatial planning
		Strategize restoration to create positive feedbacks
		Monitoring resources
Gathering data about fisheries		
	Questions to consider	
		How many people are fishing?
		What methods are they using to catch marine life?
		How often are they fishing?

Step	Goal	Supporting Actions
		Where are they fishing?
		What are they catching?
		How are laws, governance structures, and reproductive health work changing fishing pressure?
		Are they effective?
		How are changes in stressors influencing the ecosystem?

6.4.1 Short-term reductions in fishing

Fishing effort must be reduced immediately, despite the many challenges this change will present. One of the most striking findings in my thesis is the remarkable growth in fishing effort (1,800% in 50 years). Unsustainably high small-scale fishing effort is widely documented in an array of fisheries globally (Pauly and Chua 1988, Stobutzki et al. 2006, Salayo et al. 2008, Muallil et al. 2014). However, I found that the changes in spatial fishing effort were nearly a magnitude greater than that estimated from non-spatial metrics. Based on my research, a return to 1980 levels of fishing effort would require a 60% reduction in mean spatial fishing effort. This reduction may not be enough. There is evidence that Philippine fisheries reached MSY in the mid-1970s (Pauly and Chua 1988), a time when fishing effort was lower still. Despite this incredible growth in effort, reducing fishing effort in South East Asia has been nearly impossible to implement and unpopular with fishers (Pauly and Chua 1988, Martin-Smith et al. 2004, Salayo et al. 2008), in part because fishing serves an important role as a safety net for the rural poor (Béné et al. 2010).

Short-term fishing effort will be reduced if people from villages are supported to move to cities. Indeed there is evidence from Norwegian fisheries that a primary factor leading to a lower number of fishers is the growth of opportunities in cities (Béné et al. 2010). Better reproductive health can also contribute to this goal by reducing accidental pregnancies, particularly among youth. Parenthood can tie young people to villages, thereby limiting their opportunities to find livelihoods that are not tied to fishing. Moving to cities can provide exposure to new occupations, but also increases risks ranging from unsanitary living conditions, to predatory lenders, to human trafficking. Thus an urban migration program would need to provide job training – including accountability for showing up to work on time, financial education, and mentorship.

Efforts must be taken to reduce the time that people in villages spend fishing, potentially through financial programs. Where these programs exist, they can be targeted at fishing households affected by management or tied to behavioral change or participation in fisheries management (e.g. guarding MPAs). There is evidence that alternative livelihood or microfinance programs – implemented in tandem with resource management programs – can create good will in in affected communities (Wright et al. 2016). There is mixed evidence, however, that financial programs effectively reduce pressures on resources (Salafsky et al. 2001). For example, fishers who stop fishing to farm seaweed may be replaced by new fishers, or return to fishing if global commodity prices decline (Hill et al. 2012).

Limiting access to the fishery will be a challenging, yet critical component of reducing fishing effort. In developing countries, such as the Philippines, small-scale fisheries plays an important role both as a buffer that absorbs excess labor and as a safety net for the poor and vulnerable segment of rural communities (Béné et al. 2010). Thus efforts to reduce effort will require striking a careful balance between ending overexploitation of ecosystems and building adaptive capacity within fishing communities. Furthermore, access limitations have proven exceptionally difficult to implement in places, like the Philippines, where cultural traditions and social institutions to support this practice either do not exist or have eroded. Yet limiting access is the only way to ensure that those who cease fishing are not replaced by new fishers (Hill et al. 2012). Access limitations can be implemented through slow and deliberate changes including: ending reliance on the ocean as employment insurance for the jobless; defining who is authorized to fish in the Danajon Bank; documenting all full- and part-time fishers through a universal fisher registry; and strengthening village level institutions such as fishers' organizations that provide accountability around who fishes and how they fish.

6.4.2 Long-term reductions in fishing

Reducing fishing effort in the long-term will depend on slowing population growth, including through addressing the multidimensional factors that contribute to current population trends. I found that enormous increases in fishing effort (1950-2010) were driven by a rapid growth in the number of fishers. Ultimately these changes were catalyzed by the human population growth (2.3% annually) (Philippines National Statisics Office 2010). Slowing population growth is particularly urgent in the Philippines, which has the greatest number of teenage pregnancies in

Asia (UNFPA et al. 2015) – 10% of teenagers (women 15 – 19 years) are already mothers. For short term reduction in fishing effort, there is evidence that fewer children can reduce the number of men who become full time fishers (D'Agnes et al. 2010). I found antidotal evidence that supported these findings – in my research several respondents mentioned that they increased their fishing effort once they had children to support. In the long-term, reproductive health interventions can reduce population growth and increase generation time, thereby reducing the number of children who become fishers (D'Agnes et al. 2010). Population growth can be slowed by increasing wages, empowering women, and through reproductive health interventions (e.g. family planning counseling; distribution and social marketing of non-clinical methods of contraception (e.g. condoms, pill); peer education to reduce the risk of unplanned pregnancies; policy advocacy to expand access to birth control) (Nelson et al. 2006, D'Agnes et al. 2010).

6.4.3 Reducing fishing intensity

By fully implementing dormant laws supporting conservation, it will be possible – in many cases – to reduce the impacts caused by damaging practices. Intensive fishing gears – many of which are illegal under existing laws – shape coastal ecosystems in undesirable ways (e.g. reducing the cover of essential habitats, reducing species diversity, and removing individuals before they reach maturity (Mangi and Roberts 2006, Hicks and McClanahan 2012). Eliminating the use of damaging gears is a tractable step because it has potential for success. There is evidence that fishers view gear restrictions as an acceptable type of fisheries management (Salayo et al. 2008). In line with this sentiment, my research found that the proportion of fishers using these gears has stopped increasing and shown a small, although insignificant, decline under recent co-management (Chapter 2). There is, however, much work to do as destructive, active, and non-selective gears still affect much of the central Danajon Bank (57%, 88% and 91% respectively; Chapter 3).

Effectively reducing intensive gears will require developing greater accountability and self-monitoring, limiting entry, and successful enforcement. Fostering accountability and self-monitoring is key. For common pool resource users (e.g. fishers) to follow rules (e.g. ceasing use of damaging fishing gears), they must believe that other users share a credible commitment to following those rules (Ostrom 1990). These attributes can be supported by improving communication amongst fishers so that they hold each other accountable. Additionally, fishers can participate in monitoring so that they know that others are following the rules (Ostrom 1990, p 188). Fishers' organizations are fertile grounds for supporting accountability and for creating arrangements for monitoring.

Limiting entry to the fishery and eliminating the practice of short-term fishing migrations (approximately 2 week; Cebuano: *biyahe*) will also be essential to create greater compliance with existing regulations. Unknown participants, who are from outside of a system, may not know of the rules or have any sense of obligation to follow them (Ostrom 1990). Yet when rule breaking is high – including by unauthorized users – resource users are less likely to follow rules. For example, my research provides evidence that people who use *kaykay*, one highly destructive form of fishing, have high levels of participation in these short-term fishing migrations (Selgrath, unpub. data).

Enforcement is a third dimension of reducing fishing intensity. This can be accomplished by improving officer training, providing funding, and by engaging community members to report illegal activity. For example, respondents reported how in the mid-1980s (particularly 1987) the practice of Liba-Liba (Danish Seine, an illegal gear) almost stopped because of strong enforcement. This gear remained dormant until a new mayor agreed to not prosecute it in 2010). It is interesting to note that in the Philippines, there is a cultural norm that obligates enforces to ignore illicit behaviour by family and friends. With this in mind, one municipality in the Danajon Bank (outside of my study area) has prohibited officers from socially interacting with fishing communities. Enforcement is most effective when sanctions for rule breaking gradually increase in severity (Ostrom 1990).

6.4.4 Strengthening institutions to improve fishing practices

To effectively accomplish the steps described above, it will be essential to strengthen institutions that support fishers as they go through the process of changing behaviours. At the village level, fishers' organizations can help improve fisher knowledge of, and compliance with, existing legislation (Ostrom 2007). Furthermore, fishers' organizations that emphasize sustainability can create a social fence that keeps fishers from straying into the intensive and illegal practices that must be eliminated. I established that 74% of villages had fisher organizations, but only 19% of fishers were participants. This leaves a wide opportunity to improve registration and participation. The strength of municipal-level governance is essential to develop because Philippine municipalities are legally tasked with managing small-scale fisheries. Thus they have the opportunity to provide coordinated support and education to village-level fishers' organizations. To this end, municipal-level fisheries positions must be formalized and professionalized. At a regional level, there is opportunity to create institutions that cross legal boundaries to holistically consider ecosystems. In the Central Danajon Bank alone, I found that active fishers came from five municipalities and two provinces. Currently, there is little inter-government management in Danajon Bank (Pietri 2015), though it has been established elsewhere in the country (Pomeroy et al. 2010). At the national and provincial scale, it will be important for governments with higher capacity levels to support village level organizations with extension support and other resources that can foster shifts towards sustainable practices (Bavinck et al. 2005, Gelcich et al. 2010). There also may be opportunities for outside partnerships, such as with religious institutions.

6.4.5 Creating and supporting protected areas

In marine spatial planning – a spatially explicit framework for promoting biodiversity conservation and sustainable use of resources – there are opportunities to protect ecosystems using existing legislation. One aspect of marine spatial planning, marine protected areas (MPAs), are widely implemented conservation and management tools in the Philippines (Horigue et al. 2012). Some MPAs are successful (Russ and Alcala 1999, Yasué et al. 2012), but many MPAs are mere paper parks (Weeks et al. 2010). If they are to benefit oceans, existing MPAs must be fully enforced.

Existing legislation provides opportunity to expand MPAs. For example, the Philippines has goals of protecting 15% of municipal waters (1998 Fisheries Code), and 10% of coral reef areas by 2020 (Philippine Marine Sanctuary Strategy (2004)). To date, however, MPA coverage falls far short of those goals, not least because 90% of the many existing MPAs in the Philippines are < 1 km^2 (Weeks et al. 2010). Small MPAs are widespread in the Danajon Bank – I found that 70% of study villages had established MPAs between 1998 (the first year that local governments had the autonomy to establish MPAs) and 2007 (Chapter 2). Where MPAs are not explicitly included in a countries existing legislation, there can be guidance from the Convention on Biodiversity (10% protection of coastal areas by 2020). To date, 168 of 196 countries have signed this convention.

As efforts to expand MPAs move forward, maps of fishing effort and ecological features can be used as a tool to engage communities. For example, maps – such as those I produced in my research – can facilitate dialogues about the areas that would be most appropriate to protect. Maps of fishing effort can be used to identify and discuss sites that fishers depend on. Spatial and temporal modeling of ecological responses to stressors, combined with maps of key ecological features, such as habitats, can inform ecological aspects of spatial planning (Margules and Pressey 2000). For instance, corals are particularly sensitive to heavy fishing pressure near densely populated areas (Chapter 5). Based on this finding, there is evidence that setting protected areas in these locations may have greater ecological benefits than setting MPAs elsewhere. Additionally, I suggest that the northern slope of the Caubian Reef would be a high priority for protection because that outer reef slope supports relatively high coral cover, but has been subject to increasing fishing pressure (Chapters 3, 4). Since I documented that total fishing effort has a lagged effect on corals (Chapter 5), the corals in the northern Caubian reef slopes may be at risk of future declines.

6.4.6 Fostering resilient traits of ecosystem

Conservation and management can support ecosystem functioning by enhancing the capacity of ecosystems to absorb disturbances. For example, I documented that living corals were more likely to be present in places with compact habitat patches and/or in deeper areas (Chapter 5). Seascape features were a critical component of restoration in Australia. Due to positive feedbacks, coastal MPAs were more effective at supporting the structure and functioning of coral reefs when the MPAs were located near mangroves (Olds et al. 2012a). The ecological benefits of these biophysical traits can be enhanced by including them in marine spatial planning. For example, new protected areas could prioritize protecting relatively deep reefs supporting compact coral patches. Additionally, efforts to restore degraded habitats (e.g. replanting mangroves) can prioritize locations that will gain additional benefits through positive feedbacks (Suding et al. 2004). Communities and researchers can be engaged in this process to monitor resources and develop a better understanding of how coastal ecosystems respond to stress.

6.4.7 Gathering data about fisheries access and participation, number of fishers

Implementing the steps described above will be more efficient and effective with a solid understanding of pressures on the systems. Using methods described in my dissertation, it will be essential to document several aspects of the fishery. This includes questions such as: How many people are fishing? What methods are they using to catch marine life? How often are they fishing? Where are they fishing? What are they catching? How are laws, governance structures, and reproductive health work changing fishing pressure? Are they effective? How are changes in stressors influencing the ecosystem? Data collection can occur through formalized processes, but should not be limited to these approaches. Municipal governments could require fishers to participate in fishers' organizations and use these organizations as a tool to collect data. For instance, municipal governments could incentivize membership in fisher organizations by offering free boat registration to members. Boat registration is mandatory in theory, but in practice is an ignored obligation.

Meeting conservation targets and limiting the number of fishers will become easier with improved knowledge about fisheries, even though I certainly know enough to get going (Johannes 1998). Historic perspectives can allow conservation programs to set appropriate targets and to plan for the types of change that a system can accommodate (Bavinck et al. 2005). Furthermore, characterizing past changes can identify management interventions that might succeed and targets for where to scale back (Jackson and Hobbs 2009).

6.4.8 Future research

From my thesis, I identify several areas for further research to support understanding the long-term development of extractive practices, the ecological impacts of these developments, and the interactions between extraction and other stressors. First, it is unknown if the diversification of fishing gears I documented is a unique characteristics of fisheries in the Danajon Bank or is a widespread response of small-scale fisheries to increasing competition and/or overexploitation. To date, I know of only one other study that explicitly measured the change in fishing gear diversity. This other example comes from Kenyan fisheries and present an example that contrasts my research findings. Kenyan small-scale fisheries demonstrated a reduction in gear diversity in response to overfishing (McClanahan et al. 2008). In this fishery, fishers shifted their effort into one intensive gear. There is a need to investigate the frequency of these two pathways, and what factors might cause a fishery to move towards gear diversification or homogenization.

Second, drivers of individuals' choices about which fishing gears to use remain uncertain. Here I found that despite systems-wide diversification, most fishers use less than three gears over their lifetimes and only one to two gears at a time. But it remains unknown who innovates new fishing gears and if new gears were primarily adopted by younger fishers or by older, more experienced fishers. Furthermore it has been hypothesized that individuals from fishing families (vs. new migrants) more likely to use less intensive gears (Pauly and Chua 1988), but I know of no empirical studies validating this theory. The system level changes, but individual-level consistency I documented in fishing gears occurred at the same time that there has been an erosion of traditional fishing cultures and an emergence of new market opportunities under globalization (Cinner et al. 2007, Anderson et al. 2011). There are still many questions how to foster sustainability in small-scale fisheries that have experienced these changes.

Third, pairing evaluations of changes in human activities with a timeline of governance interventions can provide a valuable opportunity to understand the effects of past governance and conservation programs on human activities and ecological conditions (Gelcich et al. 2010). In order to continue to advance conservation, it is essential to stop and take stock of what has resulted from the efforts I have put in place (Butchart et al. 2015). My research is the first to focus on the influence of governance on fishing gears and the spatial dynamics of fishing effort for small-scale fisheries. From my dissertation, it is apparent that some past governance policies and priorities influence current fishing practices while others have failed to meet their objectives. There are opportunities for future research to investigate how transitions in governance can support parallel changes in fishing activities (Gelcich et al. 2010).

Fourth, there is still a great deal to learn about the ecological effects of multiple stressors. I focused on interactions among local stressors, including fishing and population pressure. Understanding local stressors is valuable because they are actionable for conservation (Côté et al. 2016). It will also be essential to understand how local stressors interact with global change, so that synergistic effects can be offset where possible. My research demonstrates the importance of long time series, particularly for stressors that have indirect effects that accumulate slowly over time. My thesis focused on quantifying small-scale fishing, a widespread stressor that is rarely measured directly (e.g. Ban et al. 2014)). When proxies are used for stressors that are difficult to measure (e.g. population density or distance to market for fishing (Brewer et al. 2009)), it will be valuable to test the relationship between the stressor and the proxy to ensure that conclusions are valid.

To address these questions, research must evaluate social-ecological dynamics at appropriate temporal and spatial scales and using appropriate tools. Developing methods for expanding conservation research to the scale of systems has been identified as a major goal for conservation (Hughes et al. 2003). For coral reef ecosystems, there have been admirable efforts to broaden our understanding of ecological dynamics and responses to stressors. For example, this trend is seen in the increasing number of efforts to compare patterns and processes both within regions (e.g. McClanahan et al. 2011) and among distant regions (e.g. Cinner et al. 2016). However, the foundational methods in these comparisons frequently remain limited to small areas within each of the regions (e.g. transects). By evaluating how spatial patterns are influenced by the biophysical variability found within systems (e.g. depth, current, river outflow, etc.), my dissertation advances this capacity in the ocean. Furthermore, I demonstrated the effectiveness of integrating satellite mapping, LEK, and hierarchical modelling to identifying relationships among stressors and corals at the scale of an ecosystem. Going forward, these approaches can be used to monitor how reefs systems are changing under the rise and fall of interacting stressors (McClanahan et al. 2014, Côté et al. 2016). Such information can be used to develop a better understanding of ecological resilience (the capacity of systems to absorb stress and continue functioning) (Holling 1973) and to ask questions about the influence of stressors on the population dynamics of species and on the behavior of individual organisms (Côté et al. 2016).

6.4.9 Conclusion

Improving strategies for protecting the ocean will be essential as these irreplaceable ecosystems face growing threats from increasing human populations, local stressors, and global change. In my research, I created a baseline documenting what small-scale fishing looked like before oceans were so thoroughly degraded, as they are today in much of the Danajon Bank. From my research it is apparent that these fisheries remain small in scale, but that their influence on the coastal ecosystem has become too big to ignore. To protect oceans, it will be essential to slow the rate of ecosystem decline by reducing stressors – through existing regulations when possible, to improve institutions that support sustainable practices, and to support attributes that contribute to a systems' resilience. Although my dissertation focused on a case study in the central Philippines, the results of my research are relevant to evaluating stressors and conservation outcomes in other small-scale fisheries and in other social-ecological systems, particularly in other small-scale industries (e.g. farming, logging, pastoral systems). It is my hope that conservation research will continue to integrate with programs to implement action to protect the systems and structures that sustain our world.

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Appendices

Appendix A Interview questions

RECONSTRUCTING SMALL-SCALE FISHING PRACTICES
Consent Form

Respondent ID # ___________________________
Name of fisher _____________________
Barangay (BGY)_____________________
LGU ____________________________

Interviewer ________________________________
Date of Interview (mm/dd/yy) ____/____/10
Purok/Sitio ___________________________
Province ____________________________

---

1. Are you ____________________?

2. I'm________ and I am a researcher from Project Seahorse.

3. I would like to interview you for a research project about changes in fishing practices and habitats in your area. This is for a study and will become part of a paper and may help people who are working to make sure that there are enough animals in the sea.

4. We have gotten permission from the Mayor and made a courtesy call to your barangay captain.

5. If you agree to participate, everything you tell us will be confidential and when we do feedback it will be general and will not identify anyone individually.

.

6. If you want to stop the interview at any time, please tell me. That is your right and it is not a problem to stop.

7. Are you willing to participate?

   ☐ Yes   ☐ No

8. The interview will take 1-2 hours – is now a good time or should we come back at another time? If you need to leave at any time, please let us know and we can hopefully reschedule a time to come back.

9. If it is ok, we would like to tape record our discussion. That will allow us to listen to your answers again. You can stop the recording at any time by telling us that you would like to stop. Are you willing to have the interview tape recorded?
   ☐ Yes   ☐ No

10. Do you give us permission to use the information we gather from our interview today for future studies about similar topics? Your information and responses will remain anonymous.

   ☐ Yes   ☐ No

FisherID: __________

RECONSTRUCTING SMALL-SCALE FISHING PRACTICES

Interview

Respondent ID # ___________________________ Interviewer: ________________________________

Respondent Name: ________________________ Date of Interview (mm/dd/yy) ____/____/10

I. GENERAL INFORMATION

1. Name:

___________________________________ _________________________

2. Age: ___________
3. Year Born: ____________
4. Sex: • Male • Female
5. Is fishing your primary job?
   ☐ Yes ☐ No

A. Highest Education:

B. Who fished in your family?

1. ☐ a. F ☐ GF ☐ GGF
   ☐ b. M ☐ 2. M ☐ 4. M
   ☐ 3. F ☐ 5. F

Gear(s):__________________________________

____

***** *NOTE: If fishing is NOT their primary job, thank for them the time and STOP the interview.*

6. Where were you born?: ____
7. *(IF not BGY):* What year did you move to BGY: _______________ *TIMELINE*
8. What is your status: ☐ Single ☐ Married ☐ Widowed ☐ Separated
9. What year were you married: ______________ *TIMELINE*
10. Number of Children: ___________
11. What year were your children born? What are their names?
    i. Do you participate in a fishers' organization? ________________
    ii. What is the name of your organization? ___________________
    iii. What year did you become a member? __________________
    iv. Are there other organizations besides your group? ___________________

When was your oldest child born? What is their first name? _TIMELINE_

- ***Repeat question for each child***
- ***If few children, ask the questions about when they went to different school levels***

II. TIMELINE FISHING QUESTIONS

12. What age did you start fishing? What year was that? Where did begin fishing? _TIMELINE_

13. Are there years that you did not fish in Getafe because you did other work or fished elsewhere?

14. *(IF YES)* What years did you not fish in Getafe or other work? _TIMELINE_

15. Have you gone fishing with an engine? What years? _TIMELINE_

16. What kind of gear(s) do you currently fish with? _TIMELINE_
    a. What years have you used these gears?

17. Did you ever fish with any other gears in the past?
    a. What years did you use these gears?
    b. Have you ever been involved with seaweed farming?

18. For all of your fishing gears, how many days per month do you fish? How has this changed over time?

III. FISHING GROUNDS and OTHER ACTIVITIES – LOCATION QUESTIONS

Orient respondent to map, have him sit facing the ocean if possible

19. Where do you keep your boat?
    - ***BLUE dot***

20. CURRENT: What is the name of a fishing ground that you currently use in this area?
    a. Where is that fishing ground located?
    b. What area do you fish in at this fishing ground? (e.g. lawod, takot, reef crest, reef flat)
    - ***GREEN – Assign FG_ID on the map, number West to East***
    - ***Repeat until you've mapped all their current FG***

21. PAST: Are there fishing grounds that you do not use now, but that you used in past years?
    a. What is the name of one of those fishing grounds?
    b. Where is it located?

c. What area do you fish in at this fishing ground? (e.g. lawod, takot, reef crest, reef flat)

- ***RED – Assign FG_ID on the map, continuing numbering from Question 21***
- ***Repeat until you've mapped all their past FG***

22. OTHER ACTIVITIES: Have you seen the following activities in any of your fishing grounds, current or past?

- ***BLACK TICS. Assign letter(s) to the fishing grounds corresponding to the activity that they saw.***

a. Spear fishing (lantern, compressor)
b. Seine
c. Turn stone
d. Blast
e. Coral harvesting
f. Poison (aquarium)
g. Muro Ami
h. Trawling/ Liba Liba

23. OTHER ACTIVITIES Are there other areas, not including your fishing grounds, where you have seen these activities while you are fishing?

- ***BLACK - Assign FG_ID on the map, continuing numbering from Question 22***
- ***On the map, assign a letter to the fishing grounds corresponding to the activity that they saw.***

IV. FISHING GROUND QUESTIONS (Current, Past)

Ask these how the questions have changed over time, referring to the timeline as reference.

24. What is the name of this fishing ground?
25. What year did you start using this fishing ground?
26. Have there been times that you did not use this fishing ground or only used it rarely?
27. How many months per year did you go to this fishing ground?
    a. How many months in a year did you go when you first started fishing there?
    b. How has the number of months that you go there in a year changed over time?
28. How many days per month did you usually go to this fishing ground?
    a. How many days in a month did you go when you first started fishing there?
    b. How has the number of days that you go there in a month changed over time?
29. What gear(s) do you use most often in this fishing ground?
    a. How has this changed?
30. What is most common habitat at this fishing ground when you most recently fished there?
    a. What was the most common habitat when you first started fishing there?
    b. When did it change?
    c. Why did it change?

31. How many other fishers in total do you usually see on one fishing trip to this fishing ground?
32. Have you seen these activities in this FG?
    a. What years have you observed these activities?
    b. How many fishers have you seen using this gear?
    c. How has the number of fishers using this year changed over time?

V. OTHER ACTIVITIES

Check observed activities on tables for the other activity areas.

33. What years have you observed these activities? *Table*
    a. How many people have you seen doing this activity? How has the number changed over time?

FisherID: __________

VI. FISHING GROUNDS IN STUDY AREA – MOST VALUABLE[1]

34. Which FG is most important to you? FG_ID: ______________________

    a. Why is it important to you?

____
____
____
____

    b. What are the disadvantages of the FG?

____
____
____
____

VII. ALL FISHING GROUNDS[2]

MAP ALL FISHING GROUNDS

1. Draw each fishing ground (FG) → Pencil
2. Draw each past Fishing Ground → Red
3. # Each Fishing Ground

4.In what FG do you each gear?

a) Why do you use these gears in these FG?
b) If you do not use illegal gears, why not? If you do use illegal gears, why do you?

____
____
____

4a.G1: FG# ______________________
4b.G1 why: ____________________________
____
____
____

4a.G3: FG# ______________________
4b.G3 why: ____________________________
____
____
____

____
4a.G5: FG# ______________________
4b.G5 why: ____________________________
____
____
____

4a.G2: FG# ______________________
4b.G2 why:
____
____
____

[1] This set of questions was used in Selgrath et al. (2014), but not included in the dissertation.
[2] This map was added partway through interviews.

4a.G4: FG# ______________________
4b.G4 why: __________________________

4a.G6: FG# ______________________
4b.G6 why: ______________________

5. Why do you change where you fish?

VIII. HABITAT QUESTIONS

35. What <u>improves</u> the amount, type or quality of habitats?

36. Does the condition of the habitats that you fish in, such as the abundance of live corals or the density of seagrass, influence the abundance of fishes? Why?

☐ Yes ☐ No

C. How has seaweed farming influenced your fishing?

37. Does the condition of the habitats you fish in influence <u>what</u> you catch?

☐ Yes ☐ No

38. Does the condition of the habitats you fish in influence <u>how much</u> you catch?

☐ Yes ☐ No

39. Does your fishing affect habitats - such as corals and seagrass- in your fishing grounds? Why or why not?

FisherID: __________

____
____
____
____

IX. GENERAL QUESTIONS

40. What part of fishing would you most like to improve?

- ☐ Catch value
- ☐ Catch volume
- ☐ Lower distance traveled
- ☐ Time spent fishing
- ☐ Other, Specify: ______________________

____
____
____
____

Respondent ID: ______________ 3.Year Born:________ 12a. Age Started Fishing:________ 12b. Year Started Fishing: ______

				Martial Law 1972 - 1981										Bising 1982		Nitang 1984						Ruping 1990					Rosing 1995											So Leyte Mud Slide 2006					
		Marcos 10/65 - 2/86																C. Aquino 2/86 - 6/92						Ramos 7/92 - 6/98						Erap 7/98 - 1/01			GMA 1/02 - 6/10									N.A. 7/10	Notes
9	Married																																										
10	Children																																										
C1																																											
C2																																											
C3																																											
C4																																											
C5																																											
C6																																											
C7																																											
C8																																											
		1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	
13/ 14	Fishing/ Not Getafe																																										
15	Engine																																										
1 16/ 17	Gears																																										
G1																																											
G2																																											
G3																																											
G4																																											
G5																																											
G6																																											
18	Days/ Month Fishing																																										
19	Hours/Day Fishing																																										
NOTES																																											

Table 2: Current/Past FG

Fisher ID: ____________________ 23. FG #: ____________________

		Marcos 10/65 - 2/86															C. Aquino 2/86 - 6/92						Ramos 7/92 - 6/98							Erap 7/98 - 1/01			GMA 1/02 - 6/10									N.A. 7/10 -	
25	Year Started:	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	Notes
25/ 26	Years Used																																										
27	Months/ Year																																										
28	Days/ Month																																										
29	Gears (Code)																																										
30	Habitat 1st Year:																																										
31	# Other Fishers																																										
32: Other Activities (Years, # People Involved)																																											
A	Spear/ Lantern																																										
B	Baling/ Gillnet																																										
C	Lukat																																										
D	Blast																																										
E	Coral Harvest																																										
F	Poison																																										
G	Muro Ami																																										
H	Trawling LibaLiba																																										
NOTES																																											

Table 3: Other Activities

Fisher ID: ________________ FG #: ____________

		Marcos 10/65 - 2/86															C. Aquino 2/86 - 6/92						Ramos 7/92 - 6/98							Erap 7/98 - 1/01			GMA 1/02 - 6/10									N.A. 7/10 -	
		1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	Notes
33: Other Activities (Years, # People Involved)																																											
A	Spear/ Lantern																																										
B	Baling/ Gillnet																																										
C	Lukat																																										
D	Blast																																										
E	Coral Harvest																																										
F	Poison																																										
G	Muro Ami																																										
H	Trawling/ LibaLiba																																										
NOTES																																											

Appendix B Fishing gears reported by respondents

Table B.1 Detailed information about the classification of 93 specific fishing gears used by fishers in the Danajon Bank, Central Visayas, Philippines (1950-2010).

Specific fishing gear	*Cebuano variations*	*General gear classes*								*Intensive categories*			
		Hook & Line	**Nets**	**Diving**	**Traps**	**Gleaning**	**Corral**	**Blast**	**Poison**	**Destructive**	**Active**	**Non-Selective**	**Illegal**
Aquarium fishing	panamelya			X						X	X	X	1932
Aquarium fishing (poison from wild plants)	panubli; tubli; kus-kus								X	X	X	X	1932
Blast fishing	tiro							X		X	X	X	1932
Blast gatherer	dugok tiro; manugok tiro; manugukay tiro							X		X	X	X	1998
Cast net: mud shrimp	laya		X								X	X	
Fish corral	bunsod; bunsod dugok						X					X	
Gaff: Sea cucumbers	spot-spot			X						X	X		
Gillnet	mangabay; mangabay nylon		X									X	
Gillnet	pukot; pukot single		X										
Gillnet, bottomset	pukot alimango; pahabog		X									X	
Gillnet, bottomset (1-2 m high): wrasses	panglunod; pukot lunod; pukot panglabayan		X									X	1986
Gillnet, bottomset (1-2 m high, fine mesh): wrasses	pukot pangayagkag; pukot kayagkag; pukot pataan		X									X	1986

Specific fishing gear	*Cebuano variations*	*General gear classes*								*Intensive categories*			
		Hook & Line	**Nets**	**Diving**	**Traps**	**Gleaning**	**Corral**	**Blast**	**Poison**	**Destructive**	**Active**	**Non-Selective**	**Illegal**
Gillnet, bottomset (compressor)	pukot compressor		X							X	X	X	
Gillnet, bottomset: crabs	pukot lambay; panglambay		X									X	
Gillnet, bottomset: mackrel, trevally	pamante		X										
Gillnet, bottomset: mackrel, trevally	pukot anduhaw; paganduhaw		X										
Gillnet, bottomset: mackrel, trevally	pukotpanagko		X										
Gillnet, drift	palutaw		X									X	
Gillnet, encircling midwater (scaring device)	panglikos ; panglikos ; pukot likos; pukot likos-likos		X								X	X	1998
Gillnet, encircling midwater: needlefish	pukot bawo; pukot balo		X								X	X	
Gillnet, midwater: mojarras	pangsamok		X							X	X	X	
Gillnet: fish	pukot isda; pukot single or triple		X									X	
Gillnet: parrotfish	pukot karabalyas		X								?		
Gillnet: parrotfish	pukot mol-mol		X								?	X	
Gillnet: tuna	pukot tulingan		X								X	X	
Gillnet, drift	pamo; pamo anduhaw; pamo barangay; pamopanagko; pukot pamo; pukot pamomangse		X									X	
Gilnet, drift (night)	panagko		X									X	

Specific fishing gear	*Cebuano variations*	*General gear classes*								*Intensive categories*			
		Hook & Line	**Nets**	**Diving**	**Traps**	**Gleaning**	**Corral**	**Blast**	**Poison**	**Destructive**	**Active**	**Non-Selective**	**Illegal**
Gilnet, drift (night): anchovies	pukot libod; libod; panglibod		X									X	
Gilnet, encircling: squid	pangnokos; pangnokos mangabay; pukot nokos		X								X	X	
Gleaning	kinhas; kinhason; panginhas; manguha; panguha; nguha lampay; lampay-lampay onguhaog kinhason; wasay-wasay					X					X	X	
Gleaning (sharp rod)	ugsang					X				X	X	X	
Gleaning and diving	pana & kinhason			X							X	X	
Gleaning and diving: crabs	mata lambay; pangmata			X							X	X	
Gleaning and diving: sea urchins	manuyom; panuyom; tuyom			X							X	X	
Handline (1 hook)	pasol; pasul	X											
Handline (1 hook, weighted)	pasol ton-ton	X											
Handline (2 hooks)	pasol double taga	X											
Handline (3 hooks)	pasol (3 hooks)	X											
Handline (5 hooks)	pasol (5 hooks)	X											
Handline (baited with squid)	pasol lahoy	X											
Handline (bobber)	patao-patao balo	X											
Handline (lantern)	palong	X											
Handline (single, moving)?	pasol labyog	X											
Handline, multiple	pasol multiple	X											
Handline, multiple	panglak-lak; panglakoy	X											

Specific fishing gear	*Cebuano variations*	***General gear classes***								***Intensive categories***			
		Hook & Line	**Nets**	**Diving**	**Traps**	**Gleaning**	**Corral**	**Blast**	**Poison**	**Destructive**	**Active**	**Non-Selective**	**Illegal**
Handline: needlefish	pasol bawo	X											
Handline: squid	pasol nokos	X											
Hook & line: tuna	pasol tulingan	X											
Jigging	subid	X											
Jigging: octopus	subid kugita	X											
Jigging: squid	subid nokos	X											
Lift net (night)	basnig		X								X	X	1998
Longline, bottomset	palangre; palangre pasol	X										X	
Longline, bottomset	pangitang; kitang	X										X	
Net, midwater (lantern)	pukot yab-yab; yab-yab		X									X	
Net, midwater (night)	lawag		X								X	X	1998
Net, midwater (scaring)	pukot sasa; pangsasa		X								X	X	1998
Net: shrimp	pasayan		X							?	?	?	
Piled rock mounds (fish aggregation with nets & cyanide)	pangito; mangito								X	X		X	1932
Pot: crabs	sapyaw				X								
Push net (triangular), small	sud-sud		X							X	X	X	
Scoop net	sikpaw		X								X	X	
Seaweed gathering	guso washout gatherer					X					X	X	
Seine, Beach	baling		X							X	X	X	
Seine, Danish	liba-liba; ring-ring; hulbot-hulbot; kub-kub; de-ring		X							X	X	X	1998

Specific fishing gear	*Cebuano variations*	*General gear classes*								*Intensive categories*			
		Hook & Line	**Nets**	**Diving**	**Traps**	**Gleaning**	**Corral**	**Blast**	**Poison**	**Destructive**	**Active**	**Non-Selective**	**Illegal**
Seine, midwater	sahid		X								X	X	1998
Seine, roundhaul	sapyaw libod; sapyaw bulinao		X								X	X	1998
Skin diving (bare hands): fish	panicop isda; panicop			X							X	X	
Skin diving (bare hands): sea cucumber	pamat			X							X	X	
Skin diving (compressor)	buso			X							X	X	
Skin diving (crowbar and lantern)	gabii lapas; ginabii lapas; lantern; manugaay; mamanaay; manusaay; ginabii lapas; lantern fishing lapas			X						X	X	X	1998
Skin diving (crowbar): abalone	kay-kay; kapinan; lapas; panglapas; pangapinan; panarap kapinan			X						X	X	X	1998
Skin diving (scoop net and lantern): shrimp	bulit pasayan; manowo pasayan; pamasayan; panu pasayan; takyan			X							X	X	
Skin diving (spear and flashlight)	ispat; pangispat			X							X	X	
Skin diving (spear and kerosene lantern)	ganta-aw; manu; manawo; pamana; pamasayan; pana lantern			X							X	X	
Skin diving (spear)	pana; pana adlaw			X							X	X	
Skin diving (twisting corals & cyanide)	sawom-sawom								X	X	X	X	1998

Specific fishing gear	*Cebuano variations*	*General gear classes*								*Intensive categories*			
		Hook & Line	**Nets**	**Diving**	**Traps**	**Gleaning**	**Corral**	**Blast**	**Poison**	**Destructive**	**Active**	**Non-Selective**	**Illegal**
Skin diving: crabs	sarap lambay			X							X	X	
Skin diving: cuttlefish	pangubutan			X							X	X	
Skin diving: shells	panarap kinhason; sarap kinhason			X							X	X	
Skin diving: various invertebrates and small fish (shells, sea cucumber, urchin, octopus)	sarap; panarap			X							X	X	
Trammel net	pukot triple; pukot lamba-lamba		X									X	
Trammelnet	pukot danggit; pangdanggit		X									X	
Trammelnet, midwater (scaring): fish	pukot dombol; dombol; tikbong		X								X	X	1998
Trammelnet: shrimp	pukot pasayan		X									X	
Trap (poison): fish	amatong; amitong		X									X	
Trap (stones): fish	bobo; bubo isda; bantak				X								
Trap, large: fish	panggal; panggal isda				X								
Trap, large: fish	simpot				X								
Trap, small: fish	panggal kuyo-kuyo; kuyo-kuyo				X								
Trap: crab	panggal lambay				X								
Trap: mudcrabs	panggal alimango				X								
Trawling (to 50 m depth)	palakaya		X							X	X	X	1998

Appendix C Yearly information for Chapter 2 calculations

Table C.1 Detailed yearly information for Chapter 2 about sample size, demographics and estimated fishing effort for 23 fishing communities in the central Danajon Bank, Philippines (1960-2010).

Year	Sample Size	Rural Population of the Philippines[a]	Estimated No. Fishers[b]	Fishing Effort (mean)	Estimated Total Fishing Effort
1960	23	18880929	2199	262	575756
1961	26	18674944	2175	257	558975
1962	31	19232549	2240	245	549884
1963	32	19798400	2306	248	570879
1964	36	20366650	2372	249	591550
1965	39	20933280	2438	251	611500
1966	40	21492918	2503	254	635887
1967	49	22051198	2568	259	664640
1968	55	22611849	2633	254	668591
1969	62	23180096	2699	255	687723
1970	74	24586309	2863	251	719542
1971	84	24264339	2826	252	713296
1972	96	24775839	2885	254	732850
1973	105	25294409	2946	253	744356
1974	115	25819661	3007	248	745108
1975	131	27110333	3157	250	787852
1976	143	26944731	3138	245	769271
1977	154	27547758	3208	242	774836
1978	161	28161324	3280	235	769924
1979	171	28786546	3352	235	788210
1980	193	30070676	3502	236	825202
1981	205	29719960	3461	233	807713
1982	215	30006173	3494	230	804888
1983	228	30280609	3526	232	816640
1984	232	30541088	3557	232	825408
1985	242	30785260	3585	228	817676
1986	259	31014810	3612	231	832768
1987	267	31225216	3636	231	841155
1988	279	31410796	3658	232	849705
1989	290	31564922	3676	234	858942
1990	302	31207518	3634	234	850043
1991	313	32501697	3785	236	894034

Year	Sample Size	Rural Population of the Philippines[a]	Estimated No. Fishers[b]	Fishing Effort (mean)	Estimated Total Fishing Effort
1992	318	33318899	3880	233	902478
1993	323	34139102	3976	235	932378
1994	326	34968627	4072	235	956670
1995	329	35481611	4132	234	965356
1996	331	36669426	4270	234	998716
1997	338	37538981	4372	234	1021263
1998	341	38419791	4474	233	1040986
1999	347	39310350	4578	232	1062808
2000	351	39791253	4634	233	1079498
2001	358	41062588	4782	240	1147119
2002	360	41922011	4882	243	1185295
2003	363	42779872	4982	244	1214785
2004	368	43626602	5081	246	1249898
2005	369	44455690	5177	245	1267032
2006	373	45164562	5260	245	1287106
2007	376	45798100	5334	246	1310235
2008	376	46527536	5418	246	1332511
2009	377	47204166	5497	248	1362002
2010	372	47294154	5508	247	1362934

Appendix D Supporting material for Chapter 2

Table D.1 Summary of GLS model statistics for changes in the total fishing effort (fishing days per year by fishers in participating villages) allocated to gears in the central Danajon Bank, Philippines (1960-2010). Full models tested the effects of year and governance period, and their interaction on total fishing effort by each category of fishing gear. Models considered changes in the use of eight general classes of fishing gears and four pairs of intensive/non-intensive gear categories.

Model	Variable	Coefficient Effect	F.value	p.value
Four common Fishing Gears				
Net				
	Intercept	-	5173.26	< 0.001
	Year	+	386.83	< 0.001
Dive				
	Intercept	-	38.54	< 0.001
	Year	+	47.45	< 0.001
Hook				
	Intercept	-	977.98	< 0.001
	Year	+	4.20	0.05
Trap				
	Intercept	+	2520.37	< 0.001
	Year	-	1.77	0.19
	Governance era		9.18	< 0.001
	Governance era (Productivity)	+		
	Governance era (Decentralized)	-		
	Governance era (Co-management)	-		
	Year: Governance era		9.69	< 0.001
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	+		
	Year: Governance era (Co-management)	+		
Four uncommon fishing gears				
Blast				
	Intercept	+	1060.69	< 0.001
	Year	-	0.18	0.67
	Governance era		5.68	< 0.01
	Governance era (Productivity)	-		
	Governance era (Decentralized)	-		

Model	Variable	Coefficient Effect	F.value	p.value
	Governance era (Co-management)	-		
	Year: Governance era		22.05	< 0.001
	Year: Governance era (Productivity)	+		
	Year: Governance era (Decentralized)	+		
	Year: Governance era (Co-management)	+		
Poison				
	Intercept	+	481.49	< 0.001
Gleaning				
	Intercept	-	229.10	< 0.001
	Year	+	2.78	0.10
	Governance era		2.23	0.10
	Governance era (Productivity)	+		
	Governance era (Decentralized)	+		
	Governance era (Co-management)	-		
	Year: Governance era		3.65	0.02
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	-		
Corral				
	Intercept	-	977.98	< 0.001
	Year	+	4.20	0.05
	Year: Governance era (Co-management)	+		
Intensive Gears				
Destructive				
	Intercept	-	59.43	< 0.001
	Year	+	13.63	< 0.001
Active				
	Intercept	-	437.89	< 0.001
	Year	+	79.33	< 0.001
Non-selective				
	Intercept	-	90.93	< 0.001
	Year	+	52.75	< 0.001
Illegal				
	Intercept	+	64.14	< 0.001
	Year	-	44.15	< 0.001
	Governance era		28.32	< 0.001
	Governance era (Productivity)	-		
	Governance era (Decentralized)	-		
	Governance era (Co-management)	-		
	Year: Governance era		2.23	0.10

Model	Variable	Coefficient Effect	F.value	p.value
	Year: Governance era (Productivity)	+		
	Year: Governance era (Decentralized)	+		
	Year: Governance era (Co-management)	+		
Non-intensive gears				
Non-destructive				
	Intercept	-	42.72	< 0.001
	Year	+	32.04	< 0.001
Passive				
	Intercept	-	144.39	< 0.001
	Year	+	9.70	< 0.01
	Governance era			
	Governance era (Productivity)	+	0.10	0.96
	Governance era (Decentralized)	+	2.50	0.07
	Governance era (Co-management)	-		
	Year: Governance era			
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	-		
	Year: Governance era (Co-management)	-		
Selective				
	Intercept	-	159.37	< 0.001
	Year	+	4.71	0.03
Legal				
	Intercept	-	831.04	< 0.001
	Year	+	55.07	< 0.001
	Governance era		19.25	< 0.001
	Governance era (Productivity)	+		
	Governance era (Decentralized)	-		
	Governance era (Co-management)	-		
	Year: Governance era		1.81	0.16
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	+		
	Year: Governance era (Co-management)	+		

Table D.2 Summary of GLS model statistics for changes in the relative fishing effort (percentage of fishing days per year) allocated to gears in the central Danajon Bank, Philippines (1960-2010). Full models tested the effects of year and governance period, and their interaction on relative fishing effort by each category of fishing gear. Models considered changes in the use of eight general classes of fishing gears and four pairs of intensive/non-intensive gear categories.

Model	Variable		Coefficient Effect	F.value	p.value
Four common Fishing Gears					
Hook					
	Intercept		+	2343.57	< 0.001
	Year		-	58.74	< 0.001
Net					
	Intercept		+	5068.87	< 0.001
	Year		-	4.28	0.04
	Governance era			2.80	0.05
		Governance era (Productivity)	-		
		Governance era (Decentralized)	-		
		Governance era (Co-management)	-		
	Year: Governance era			6.90	< 0.001
		Year: Governance era (Productivity)	+		
		Year: Governance era (Decentralized)	+		
		Year: Governance era (Co-management)	+		
Poison					
	Intercept		+	744.30	< 0.001
	Governance era			4.50	< 0.01
		Governance era (Productivity)	+		
		Governance era (Decentralized)	+		
		Governance era (Co-management)	+		
Trap					
	Intercept		+	1379.63	< 0.001
	Year		-	23.81	< 0.001
	Governance era			4.00	0.01
		Governance era (Productivity)	+		
		Governance era (Decentralized)	-		
		Governance era (Co-management)	-		
	Year: Governance era			5.21	< 0.01

Model	Variable	Coefficient Effect	F.value	p.value
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	+		
	Year: Governance era (Co-management)	+		

Table D.3 Summary of GLS model statistics for changes in the proportion of fishers using various fishing gears in a year in the central Danajon Bank, Philippines (1960-2010). Full models tested the effects of year and governance period, and their interaction on the percentage of fishers by each category of fishing gear. Models considered changes in the use of eight general classes of fishing gears and four pairs of intensive/non-intensive gear categories.

Model	Variable	Coefficient Effect	F.value	p.value
Four common Fishing Gears				
Net				
	Intercept	-	1999.79	< 0.001
	Year	+	12.10	< 0.01
Dive				
	Intercept	-	720.71	< 0.001
	Year	+	15.19	< 0.001
	Governance era		0.48	0.70
	Governance era (Productivity)	+		
	Governance era (Decentralized)	+		
	Governance era (Co-management)	+		
	Year: Governance era		1.87	0.15
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	-		
	Year: Governance era (Co-management)	-		
Hook				
	Intercept	-	6184.29	< 0.001
	Year	+	27.01	< 0.001
	Governance era		2.04	0.12
	Governance era (Productivity)	+		
	Governance era (Decentralized)	+		
	Governance era (Co-management)	+		
	Year: Governance era		8.06	< 0.001
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	-		
	Year: Governance era (Co-management)	-		
Trap				
	Intercept	+	715.12	< 0.001

Four uncommon fishing gears				
Blast				
	Intercept	+	300.77	< 0.001
	Year	-	2.96	0.09
	Governance era		0.07	0.98
	Governance era (Productivity)	-		
	Governance era (Decentralized)	-		
	Governance era (Co-management)	-		
	Year: Governance era		6.63	< 0.001
	Year: Governance era (Productivity)	+		
	Year: Governance era (Decentralized)	+		
	Year: Governance era (Co-management)	+		
Poison				
	Intercept	-	102.95	< 0.001
	Year	+	4.20	0.045912
Gleaning				
	Intercept	+	306.90	< 0.001
Corral				
	Intercept	-	6184.29	< 0.001
	Year	+	27.01	< 0.001
	Governance era		2.04	0.12
	Governance era (Productivity)	+		
	Governance era (Decentralized)	+		
	Governance era (Co-management)	+		
	Year: Governance era		8.06	< 0.001
	Year: Governance era (Productivity)	-		
	Year: Governance era (Decentralized)	-		
	Year: Governance era (Co-management)	-		
Intensive Gears				
Destructive				
	Intercept	+	469.06	< 0.001
Active				
	Intercept	+	2057.85	< 0.001
	Year	-	3.65	0.06
	Governance era		0.33	0.81
	Governance era (Productivity)	-		
	Governance era (Decentralized)	-		
	Governance era (Co-management)	-		
	Year: Governance era		2.56	0.07
	Year: Governance era (Productivity)	+		
	Year: Governance era (Decentralized)	+		

Year: Governance era (Co-management)	+		
NonSelective			
Intercept	-	2594.48	< 0.001
Year	+	22.78	< 0.001
Illegal			
Intercept	+	492.85	< 0.001
Year	-	378.33	< 0.001
Governance era		148.44	< 0.001
Governance era (Productivity)	-		
Governance era (Decentralized)	-		
Governance era (Co-management)	-		
Year: Governance era		7.49	< 0.001
Year: Governance era (Productivity)	+		
Year: Governance era (Decentralized)	+		
Year: Governance era (Co-management)	+		
Non-intensive gears			
NonDestructive			
Intercept	-	38997.43	< 0.001
Year	+	0.17	0.685846
Governance era		5.90	< 0.01
Governance era (Productivity)	+		
Governance era (Decentralized)	+		
Governance era (Co-management)	-		
Year: Governance era		4.45	< 0.01
Year: Governance era (Productivity)	-		
Year: Governance era (Decentralized)	-		
Year: Governance era (Co-management)	+		
Passive			
Intercept	+	1833.84	< 0.001
Year	-	0.07	0.785763
Governance era		1.29	0.291541
Governance era (Productivity)	+		
Governance era (Decentralized)	+		
Governance era (Co-management)	-		
Year: Governance era		3.27	0.030295
Year: Governance era (Productivity)	-		
Year: Governance era (Decentralized)	-		
Year: Governance era (Co-management)	+		
Selective			
Intercept	+	1304.21	< 0.001
Legal			

Intercept	-	753180619.02	< 0.001
Year	+	1386.13	< 0.001
Governance era		36.73	< 0.001
Governance era (Productivity)	+		
Governance era (Decentralized)	+		
Governance era (Co-management)	+		
Year: Governance era		5.54	< 0.01
Year: Governance era (Productivity)	-		
Year: Governance era (Decentralized)	-		
Year: Governance era (Co-management)	-		

Appendix E Supporting material for Chapter 3

E.1 Methods: Additional details

Study site

Our study area spanned five municipalities, two provinces, and a gradient from inshore turbid waters to offshore clear waters. Changes in the spatial location of fishing can be described by the geomorphic and ecological characteristics of the locations where the fishing occurs. Coral reefs can be divided into geomorphic zones including: reef flat,; reef slope; channels between reefs and/or islands. The Danajon Bank supports three ecological zones (modified from Hansen et al. 2011): (1) a coastal zone of turbid waters, extensive mangroves, and terrestrial islands that support farming; (2) an outer reef zone with clear waters, extensive seagrass beds, and scattered cays; and (3) an inner reef zone with intermediate characteristics of the coastal and outer reef areas. Through ongoing partnerships, Project Seahorse has worked in this region for 20 years, including establishing MPAs with several communities participating in this study.

Estimating demographic changes

We used demographic data and information collected during interviews to estimate past demographic changes. We considered fisher demographics for two reasons. First, we wanted to quantify community-specific patterns of fishing. Second, we used this community-specific information to estimate the number of people who fished in the study area during past decades (see Figure 3.2 for an overview of data sources and calculations). We gathered demographic information from municipal offices, community census records, and during interviews. Throughout the Philippines, census records are collected regularly by community health workers.

Where occupational information in the census was incomplete, unclear, or had questionable accuracy (e.g. 27 year olds listed as high school students), we confirmed individuals' occupations with the village health workers. We used village census records rather than municipal records of fishers because we found that municipal data was inaccurate and severely underestimated the number of fishers. In one of the municipalities, for example, official records listed 85 fishers, yet we documented > 1000 fishers. Census occupational data provided a conservative estimate of fishers because it did not include children or women who fish (Kleiber et al. 2014). Thus we assume we underestimated total fishing effort, particularly in inter-tidal areas where women's fisheries are focused (Kleiber et al. 2014).

We were unable to obtain detailed information about village demographics in past decades. The Philippines census office aggregates data at the municipal level and does not distinguish between inland and coastal communities. Thus for past years, we calculated estimates of the fisher population size and the number of fishers targeting the study area using three assumptions: that village populations changed at the same rate as the rural population of the Philippines, that the proportion of village residents who fished was consistent over time, and that the proportion of people who fished in the mapped area was consistent over time. We based this estimate on the approach from Teh and Sumaila (2013) which estimates the number of fishers by multiplying an identified proportion of people who fish by the coastal population. We expanded this method for estimating the number of fishers back in time. We recognize that demographic changes differ among communities due to factors such as variable population growth, livelihood opportunities, and migration patterns. Since this detailed information was not available, our methods provide the best estimation of changing populations in fishing communities.

Sampling and fisher interviews

The LEK of small-scale fishers includes the integrated and situated knowledge, practices, and beliefs of regarding the local environment (Berkes 2012, McMillen et al. 2014). Here, we use one LEK approach, participatory mapping. During participatory mapping fishers' create maps of spatial fishing dynamics over time, including spatial extent and gear use (Neis et al. 1999). To create participatory maps, we sampled 50% of fishing communities in the study area. We also sampled communities up to 10 km away from the study area (Figure 3.1).

Since we were interested in a long time series, we stratified the fishers by age prior to random sampling. We focused our interviews on fishers who were born before 1981 because we estimated that most of this age group would have fished for at least 15 years. This method may underrepresent recent trends driven by young fishers, but prioritizes long time-series of fishing. Interviews were conducted in the local language (Cebuano) by two local research assistants. Most interviews were conducted in respondent's houses or yards. Some interviews were private (only the respondent and the research assistant), while household members were present during other interviews. The setting was based on the respondent's preference. If a randomly selected fisher was unavailable, we either substituted other household members for the original respondent (i.e. if another household members also fished) or substituted the fisher with someone from a randomly selected backup list we made for this purpose.

During interviews we spent a long time building personal relationships with the respondents, which was the most effective way of getting them to talk openly about both legal and illegal things they have done (Chambers 1994). We felt was very important given that we suspected a large proportion of fishers fished in illegal areas and used illegal gears, which are difficult to document accurately (Gavin et al. 2010).

Standardizing fishing grounds

To evaluate how fishing effort changed over time, we synthesized information from the fishing timelines and maps using five steps. Details of these steps are provided in the main body of the manuscript, and additional details about standardizing are included here. Following digitizing, we standardized the widths of the fishing ground polygons that were located in deep areas. We decided this step was needed because deep fishing ground polygons had inconsistent widths (e.g. skinny vs. wide) based on which research assistant had done the interview. Standardizing was not necessary for shallow fishing ground polygons. In shallow areas, polygons were clearly aligned with the geographic features that were visible in the satellite images. We set the width of deep fishing grounds at 2/3 of the channel width, centered on the original polygon. After summing the maps, we found that there were some small gaps where no fishing was mapped between fishing grounds. As these were likely due to mapping errors, we smoothed maps by merging gaps < 10 ha with neighboring polygons. Most gaps were small, with approximately 95% of gaps < 0.5 ha. Calculations were done in in ArcGIS 10.2 (Environmental Systems Research Institute, Redlands, California).

E.2 Discussion

This study highlights the importance of including space when evaluating developments in small-scale fisheries, including trends in effort and CPUE. Fishing effort remains one of the most poorly documented aspects of marine fisheries (Anticamara et al. 2011), yet is vital for understanding how fishing pressure and CPUE have changed (Walters 2003). We found that increases in mean fishing effort were much greater when space was considered. This disparity parallels the disparity between spatial vs. non-spatial differences from global studies of industrial fishing, where estimates of effort growth (1950-2010) were approximately ten-times greater when space was included (Anticamara et al. 2011, Watson et al. 2012).

What effort levels can be supported?

Unsustainably high small-scale fishing effort is widely documented in an array of fisheries globally (Pauly and Chua 1988, Stobutzki et al. 2006, Worm et al. 2006, Salayo et al. 2008, Muallil et al. 2014). Historical records of fishing, such as we identified, can be used to set targets for reducing fishing levels (Jackson and Hobbs 2009). The Danajon Bank fisheries require a 60% reduction in mean spatial fishing effort to scale back to 1980 levels of effort. Our 60% estimate is roughly in line with modeling of small-scale fisheries in the Philippines which identified that 53% of fishers needed to exit or that up to 59% of waters needed to be protected in no take MPAs to avert collapse (Muallil et al. 2014). Although these substantial targets (60% and 53%) are not immediately achievable, it is necessary to begin implementing programs to reducing effort immediately. These efforts will need to address challenges such as the rapidly growing human population (National Statistical Coordination Board 2010), the open access nature of the fishery (Ostrom 1990), and the important roles small-scale fisheries plays as a buffer and safety net for rural communities (Béné et al. 2010). Thus reducing effort in small-scale fisheries will require parallel efforts to foster adaptive capacity within fishing communities (FAO 2015).

Table E.1 Changes in the frequency and distribution of fishing effort (days per year) at specific locations (grid cells) in the Danajon Bank ecosystem (Central Visayas, Philippines) for three subsets of fishing gears: (a) All fishing; (b) Fishing gear classes; and (c) Intensive fishing gear categories.

		Paired-Sample t Test			Kolmogorov-Smirnov Test	
Map	**Years**	**t**	**df**	**p-value**	**D**	**p-value**
(a) All fishing						
	1960-1970	-19.94	999	< 0.001	0.45	<0.001
	1970-1980	-29.68	999	< 0.001	0.41	<0.001
	1980-1990	-26.19	999	< 0.001	0.24	<0.001
	1990-2000	6.98	999	1	0.06	<0.001
	2000-2010	8.11	999	1	0.08	<0.001
(b) Fishing gear classes						
Hook-and-line	1960 - 1970	-16.45	999	<0.001	0.28	<0.001
	1970 - 1980	-18.87	999	<0.001	0.27	<0.001
	1980 - 1990	-22.64	999	<0.001	0.20	<0.001
	1990 - 2000	3.44	999	1.00	0.07	<0.001
	2000 - 2010	0.96	999	0.83	0.10	<0.001
Net	1960 - 1970	-13.9	999	<0.001	0.30	<0.001
	1970 - 1980	-23.44	999	<0.001	0.38	<0.001
	1980 - 1990	-21.75	999	<0.001	0.24	<0.001
	1990 - 2000	2.47	999	0.99	0.09	<0.001
	2000 - 2010	17.66	999	1	0.10	<0.001
Diving	1960 - 1970	-7.67	999	<0.001	0.08	<0.001
	1970 - 1980	-11.14	999	<0.001	0.11	<0.001
	1980 - 1990	-11.6	999	<0.001	0.14	<0.001
	1990 - 2000	-8.87	999	<0.001	0.03	<0.001
	2000 - 2010	-9.16	999	<0.001	0.11	<0.001
Trap	1960 - 1970	-5.42	999	<0.001	0.07	<0.001
	1970 - 1980	-2.45	999	<0.01	0.05	<0.001

		Paired-Sample t Test			Kolmogorov-Smirnov Test	
Map	**Years**	**t**	**df**	**p-value**	**D**	**p-value**
	1980 - 1990	2.83	999	1.00	0.02	<0.001
	1990 - 2000	-1.3	999	0.10	0.01	<0.01
Blast	1960 - 1970	1.85	999	0.97	0.01	0.55
	1970 - 1980	-11.37	999	<0.001	0.13	<0.001
	1980 - 1990	9.24	999	1	0.11	<0.001
	1990 - 2000	-1.44	999	0.07	0.02	<0.01
	2000 - 2010	-3.36	999	<0.001	0.01	0.11
Poison	1960 - 1970	-5.06	999	<0.001	0.02	<0.001
	1970 - 1980	-2.19	999	0.01	0.03	<0.001
	1980 - 1990	1.2	999	0.88	0.01	<0.01
	1990 - 2000	-4.49	999	<0.001	0.02	<0.001
	2000 - 2010	-1.43	999	0.08	0.02	<0.001
Glean	1960 - 1970	-4.88	999	<0.001	0.02	<0.001
	1970 - 1980	0.45	999	0.67	0.02	<0.001
	1980 - 1990	-4.35	999	<0.001	0.03	<0.001
	1990 - 2000	-3.29	999	<0.001	0.02	<0.001
	2000 - 2010	-1.46	999	0.07	0.01	<0.01
Corral	1970 - 1980	-1.73	999	0.04	0.00	1
	1980 - 1990	-0.71	999	0.24	0.00	1
	1990 - 2000	-1.4	999	0.08	0.00	0.9
	2000 - 2010	-1.09	999	0.14	0.00	0.9
(c) Intensive fishing gear categories						
Destructive	1960 - 1970	-1.01	999	0.16	0.12	<0.001
	1970 - 1980	-15.74	999	<0.001	0.25	<0.001
	1980 - 1990	-18.24	999	<0.001	0.28	<0.001
	1990 - 2000	-7.34	999	<0.001	0.09	<0.001
	2000 - 2010	10.4	999	1	0.11	<0.001
Non-destructive	1960 - 1970	-20.08	999	<0.001	0.45	<0.001
	1970 - 1980	-30.19	999	<0.001	0.38	<0.001
	1980 - 1990	-26.9	999	<0.001	0.23	<0.001
	1990 - 2000	13.13	999	1	0.09	<0.001
	2000 - 2010	4.24	999	1	0.05	<0.001
Active	1960 - 1970	-4.39	999	<0.001	0.29	<0.001
	1970 - 1980	-19.52	999	<0.001	0.33	<0.001
	1980 - 1990	-24.21	999	<0.001	0.31	<0.001
	1990 - 2000	-9.07	999	<0.001	0.07	<0.001
	2000 - 2010	11	999	1	0.14	<0.001
Passive	1960 - 1970	-19.62	999	<0.001	0.45	<0.001
	1970 - 1980	-28.46	999	<0.001	0.35	<0.001
	1980 - 1990	-23.9	999	<0.001	0.22	<0.001
	1990 - 2000	15.16	999	1	0.12	<0.001
	2000 - 2010	-0.65	999	0.26	0.05	<0.001
Non-selective	1960 - 1970	-8.44	999	<0.001	0.36	<0.001

Map	Years	Paired-Sample t Test			Kolmogorov-Smirnov Test	
		t	**df**	**p-value**	**D**	**p-value**
	1970 - 1980	-25.66	999	<0.001	0.40	<0.001
	1980 - 1990	-26.8	999	<0.001	0.25	<0.001
	1990 - 2000	-3.08	999	<0.01	0.05	<0.001
	2000 - 2010	10.97	999	1	0.11	<0.001
Selective	1960 - 1970	-17.73	999	<0.001	0.36	<0.001
	1970 - 1980	-20.32	999	<0.001	0.40	<0.001
	1980 - 1990	-20.65	999	<0.001	0.25	<0.001
	1990 - 2000	11.2	999	1	0.05	<0.001
	2000 - 2010	-2.58	999	<0.01	0.11	<0.001
Illegal	1960 - 1970	5.15	999	1	0.02	<0.001
	1970 - 1980	-11.34	999	<0.001	0.13	<0.001
	1980 - 1990	-3.84	999	<0.001	0.11	<0.001
	1990 - 2000	-19.35	999	<0.001	0.44	<0.001
	2000 - 2010	11.53	999	1	0.11	<0.001
Legal	1960 - 1970	-21.58	999	<0.001	0.46	<0.001
	1970 - 1980	-31.12	999	<0.001	0.41	<0.001
	1980 - 1990	-25.69	999	<0.001	0.27	<0.001
	1990 - 2000	20.49	999	1	0.18	<0.001
	2000 - 2010	2.45	999	0.99	0.06	<0.001

Note: t tests were calculated using fisher densities at 1000 randomly distributed points. Kolmogorov-Smirnov tests were calculated using the values from all grid cells.

Table E.2 Changes in the extent and effort (days per year) at specific locations (grid cells) of small-scale fishing in the Danajon Bank Ecosystem (Central Visayas, Philippines) for classes of fishing gears.

Map	Year	% Ocean Fished	Cumulative Effort (Mean (SD))	Cumulative Effort (Max.)
Hook-and-line	1960	9%	4 (13)	168
	1970	32%	37 (70)	312
	1980	58%	125 (175)	750
	1990	67%	253 (296)	1,338
	2000	70%	239 (273)	1,327
	2010	70%	237 (236)	1,118
Net	1960	5%	5 (26)	180
	1970	35%	32 (60)	228
	1980	56%	243 (332)	1,405
	1990	66%	592 (768)	3,455
	2000	70%	577 (732)	3,086
	2010	70%	414 (487)	2,062
Diving	1960	1%	1 (6)	60
	1970	9%	4 (18)	204
	1980	20%	29 (86)	883
	1990	34%	46 (102)	974
	2000	36%	58 (129)	1,288
	2010	46%	84 (201)	1,869
Trap	1960	3%	8 (50)	546
	1970	11%	16 (50)	420
	1980	15%	21 (70)	756
	1990	16%	20 (64)	663
	2000	16%	20 (65)	496
	2010	35%	57 (104)	526
Blast	1960	1%	1 (12)	228
	1970	1%	0 (1)	19
	1980	14%	36 (98)	311
	1990	19%	15 (35)	283
	2000	20%	16 (36)	312
	2010	19%	16 (37)	336
Poison	1960	1%	0 (1)	6
	1970	3%	1 (6)	44

Map	Year	% Ocean Fished	Cumulative Effort (Mean (SD))	Cumulative Effort (Max.)
	1980	3%	2 (10)	89
	1990	2%	1 (8)	56
	2000	2%	1 (9)	64
	2010	2%	2 (15)	372
Glean	1960	1%	2 (14)	120
	1970	3%	3 (16)	164
	1980	3%	3 (16)	202
	1990	6%	4 (21)	271
	2000	7%	5 (23)	279
	2010	7%	8 (42)	636
Corral	1960	0.01%	0 (4)	312
	1970	0.03%	0 (5)	312
	1980	0.3%	0 (5)	310
	1990	0.3%	0 (9)	239
	2000	0.4%	1 (18)	287
	2010	0.4%	1 (19)	312

Table E.3 Changes in the extent and effort (days per year) at specific locations (grid cells) of small-scale fishing in the Danajon Bank Ecosystem (Central Visayas, Philippines) for categories of intensive fishing gears and their non-intensive counterparts.

Map	Year	% Ocean Fished	Cumulative Effort (Mean (SD))	Cumulative Effort (Max.)
Destructive	1960	4%	4 (25)	228
	1970	15%	6 (15)	140
	1980	38%	62 (115)	470
	1990	58%	185 (270)	1,083
	2000	59%	246 (405)	1,650
	2010	57%	157 (214)	1,104
Non-destructive	1960	20%	48 (119)	648
	1970	65%	146 (205)	916
	1980	89%	486 (477)	2,218
	1990	92%	921 (887)	4,471
	2000	92%	806 (717)	3,594
	2010	93%	769 (638)	3,494
Active	1960	20%	23 (55)	480
	1970	48%	28 (48)	324
	1980	67%	173 (251)	1,489
	1990	86%	373 (435)	2,002
	2000	88%	450 (565)	2,723
	2010	88%	315 (328)	2,573
Passive	1960	10%	29 (95)	648
	1970	55%	124 (182)	744
	1980	84%	377 (397)	1,691
	1990	87%	733 (762)	3,718
	2000	87%	602 (578)	2,932
	2010	88%	612 (579)	2,697
Non-selective	1960	20%	39 (105)	792
	1970	55%	59 (86)	516
	1980	78%	319 (381)	1,750
	1990	90%	661 (677)	3,384
	2000	91%	696 (733)	3,452
	2010	91%	553 (484)	3,070
Selective	1960	5%	12 (67)	648

Map	Year	% Ocean Fished	Cumulative Effort (Mean (SD))	Cumulative Effort (Max.)
	1970	50%	93 (136)	576
	1980	75%	230 (243)	1,572
	1990	75%	445 (496)	2,480
	2000	78%	356 (365)	1,932
	2010	81%	373 (366)	1,708
Illegal	1960	3%	4 (25)	186
	1970	3%	0 (1)	8
	1980	14%	36 (98)	333
	1990	21%	46 (112)	486
	2000	65%	299 (467)	2,134
	2010	63%	193 (255)	1,428
Legal	1960	21%	48 (119)	696
	1970	66%	151 (205)	960
	1980	90%	511 (481)	2,361
	1990	92%	1058 (1036)	5,068
	2000	92%	753 (644)	3,377
	2010	93%	734 (594)	3,311

Appendix F Supporting materials for Chapter 3

Table F.1 Descriptions, classifications, and categories of fishing gears used by those respondents who fished in the section of the central Danajon Bank mapped in Chapter 3 (1960 - 2010).

Fishing gear	**Main gear class**									**Category**			
	Hook-and-line	Nets	Diving	Traps	Gleaning	Corral	Blast	Poison		Active	Destructive	Non-selective	Illegal
Aquarium fishing			X							X	X	X	1932
Aquarium fishing (poison)								X		X	X	X	1932
Blast fishing							X			X	X	X	1932
Blast gatherer							X			X	X	X	1998
Cast net: mud shrimp		X								X		X	
Fish corral						X						X	
Gaff: Sea cucumbers			X							X	X		
Gillnet1		X											
Gillnet2		X										X	
Gillnet, bottom-set		X										X	
Gillnet, bottom-set (1-2 m high): wrasses		X										X	
Gillnet, bottom-set (1-2 m high, fine mesh): wrasses		X										X	1986
Gillnet, bottom-set (compressor)		X								X	X	X	
Gillnet, bottom-set: coral reef fishes		X											

Fishing gear	Main gear class									Category			
	Hook-and-line	Nets	Diving	Traps	Gleaning	Corral	Blast	Poison		Active	Destructive	Non-selective	Illegal
Gillnet, bottom-set: crabs		X										X	
Gillnet, bottom-set: mackerel, trevally		X											
Gillnet, drift		X										X	
Gillnet, encircling midwater (scaring device)		X								X		X	1998
Gillnet, encircling midwater: needlefish		X								X		X	
Gillnet, midwater: mojarras		X								X	X	X	
Gillnet: fish		X										X	
Gillnet1: parrotfish		X											
Gillnet2: parrotfish		X										X	
Gillnet: tuna		X								X		X	
Gillnet, drift		X										X	
Gillnet, drift (night)		X										X	
Gillnet, drift (night): anchovies		X										X	
Gillnet, encircling: squid		X								X		X	
Gleaning					X					X		X	
Gleaning (sharp rod)					X					X	X	X	

Fishing gear	Main gear class									Category			
	Hook-and-line	Nets	Diving	Traps	Gleaning	Corral	Blast	Poison		Active	Destructive	Non-selective	Illegal
Gleaning and diving			X							X		X	
Gleaning and diving: crabs			X							X		X	
Gleaning and diving: sea urchins			X							X		X	
Handline (1 hook)	X												
Handline (1 hook, weighted)	X												
Handline (2 hooks)	X												
Handline (3 hooks)	X												
Handline (5 hooks)	X												
Handline (baited with squid)	X												
Handline (bobber)	X												
Handline (lantern)	X												
Handline (single, moving)	X												
Handline, multiple	X												
Handline: needlefish	X												
Handline: squid	X												
Hook & line: tuna	X												
Jigging	X												
Jigging: octopus	X												
Jigging: squid	X												

Fishing gear	Main gear class									Category			
	Hook-and-line	Nets	Diving	Traps	Gleaning	Corral	Blast	Poison		Active	Destructive	Non-selective	Illegal
Lift net (night)		X								X		X	1998
Longline, bottom-set	X											X	
Net, midwater (lantern)		X										X	
Net, midwater (night)		X								X		X	1998
Net, midwater (scaring)		X								X		X	1998
Net: shrimp		X								X		X	
Piled rock mounds (fish aggregation with nets & cyanide)								X			X	X	1932
Pot: crabs													
Scissor net (night): needlefish		X								X	X	X	
Scoop net		X								X		X	
Seaweed gathering					X					X		X	
Seine, beach		X								X	X	X	
Seine, Danish		X								X	X	X	1998
Seine, midwater		X								X		X	1998
Seine, round haul		X								X		X	1998
Skin diving (bare hands): fish			X							X		X	
Skin diving (bare hands): sea cucumber			X							X		X	

Fishing gear	**Main gear class**									**Category**			
	Hook-and-line	Nets	Diving	Traps	Gleaning	Corral	Blast	Poison		Active	Destructive	Non-selective	Illegal
Skin diving (compressor)			X							X		X	
Skin diving (crowbar and lantern)			X							X	X	X	1998
Skin diving (crowbar): abalone			X							X	X	X	1998
Skin diving (scoop net and lantern): shrimp			X							X		X	
Skin diving (spear and flashlight)			X							X		X	
Skin diving (spear and kerosene lantern)			X							X		X	
Skin diving (spear)			X							X		X	
Skin diving (twisting corals & cyanide)								X		X	X	X	1998
Skin diving: crabs			X							X		X	
Skin diving: cuttlefish			X							X		X	
Skin diving: shells			X							X		X	
Skin diving: various invertebrates and small fish			X							X		X	
Trammel net		X										X	

Fishing gear	Main gear class									Category			
	Hook-and-line	Nets	Diving	Traps	Gleaning	Corral	Blast	Poison		Active	Destructive	Non-selective	Illegal
Trammel net, midwater (scaring): fish		X								X		X	1998
Trammel net: shrimp		X										X	
Trap (poison): fish		X										X	
Trap (stones): fish													
Trap, large: fish													
Trap, small: fish													
Trap: crab													
Trap: mud crabs													
Trawling (to 50 m depth)		X								X	X	X	1998

Appendix G Supporting materials for Chapter 4

Table G.1 Habitat classes used in the local environmental knowledge (LEK) mapping approach, and major habitat classes used for the map comparison.

Benthic Habitats	Major Habitat Classes
Coral	Coral
Coral/rubble	
Coral/rubble/sand	
Coral/rubble/sand/seagrass	
Coral/rubble/seagrass	
Coral/sand	
Coral/seagrass	
Rubble	Rubble
Rubble/sand/seagrass	
Rubble/seagrass	
Algae	
Sand	Sand
Sand/seagrass	Seagrass
Seagrass	
Deep water	Not analyzed (deep water, land, mangroves, not mapped) †
Land	
Mangrove	
Not mapped	

†Excluded from analyses. We excluded Mangroves from comparisons because the LEK mapping method used in this study focused solely on mapping shallow fishing grounds that were rarely located in mangrove habitats.

Table G.2 Hierarchical habitat classes used for the remote sensing (RS) mapping approach, and major benthic habitat classes used for the map comparison.

Reef Hierarchy	Geomorphic Hierarchy	Benthic Hierarchy	Major Habitat Classes
Reef	Reef flat inner	Coral/algae rubble	Coral
	Reef flat outer	Rubble coral	Rubble
	Reef slope	Rubble coral/algae	
		Sand rubble coral	
		Sand rubble seagrass	
		Sand	Sand
		Sand, terrestrial	
		Sand seagrass	Seagrass
		Sand seagrass/algae	
		Seagrass, dense	
		Seagrass, medium	
Cloud	Cloud	Cloud	Not analyzed (deep water, land, mangroves, cloud) †
Deep water	Deep water	Deep water	
Land	Cay	Land	
	Terrestrial island		
	Mainland		
	Mangrove	Mangrove	
	Mangrove re-vegetation	Mangrove re-vegetation	

Note: for the benthic hierarchy, a space distinguishes dominant and secondary habitats while a "/" distinguishes co-dominant habitats.

†Excluded from analyses. We excluded Mangroves from comparisons because the LEK mapping method used in this study focused solely on mapping shallow fishing grounds that were rarely located in mangrove habitats.

Table G.3 Comparative agreement between maps made using local environmental knowledge (LEK) and remote sensing (RS) habitat mapping approaches at 1,000 random points. (a) Map comparison using major habitat classes that correspond to the validations data. (b) Map comparison using habitat classes from the original LEK and RS maps.

(a)

RS Map	**LEK Map Habitats**				**Total**	**Agreement**
	Coral	**Rubble**	**Sand**	**Seagrass**		**LEK to RS**
Habitats						
Coral	**30**	35	3	2	70	43%
Rubble	76	**193**	4	44	317	61%
Sand	97	48	**15**	22	182	8%
Seagrass	134	156	14	**127**	431	29%
Total	337	432	36	195	1,000	
Agreement						
RS to LEK	9%	45%	42%	65%		
Overall						37%

(b)

	LEK Map Habitats														Agreement
RS Map	**Co**	**Co/ Ru**	**Co/ Sa**	**Co/ Sg**	**Co/ Ru/ Sa**	**Co/ Ru/ Sg**	**Co/ Ru/ Sa/ Sg**	**Ru**	**Ru/ Sg**	**Ru/ Sa/ Sg**	**Sa**	**Sa/ Sg**	**Sg**	**Total**	**LEK to RS**
Habitats															
Co/Ag Ru	**12**	**38**	**0**	**0**	**5**	**3**	**0**	**7**	**0**	**2**	1	1	1	70	96%
Ru Co	**7**	**24**	**0**	**1**	**1**	**0**	**3**	**6**	**0**	**1**	1	0	0	44	98%
Ru Co/Ag	**6**	**22**	**6**	**8**	**2**	**13**	**10**	**38**	**3**	**19**	2	7	12	148	86%
Sa	15	28	**7**	34	**22**	6	**4**	6	1	**11**	**5**	**0**	9	148	33%
Sa, Terr	4	2	**0**	2	**3**	1	**2**	6	0	**1**	**9**	**1**	3	34	47%
Sa Ag/Sg	0	0	**0**	**0**	0	**0**	**1**	1	**0**	**0**	**0**	**0**	**0**	2	50%
Sa Ru Co	**3**	**39**	**3**	**0**	**7**	**2**	**2**	**5**	**0**	**4**	**0**	**0**	0	65	100%
Sa Ru Sg	1	**27**	**0**	**10**	**2**	**3**	**3**	**3**	**1**	**10**	**0**	**0**	**0**	60	98%
Sa Sg	16	39	16	**27**	33	**23**	**10**	27	**18**	**21**	**8**	**13**	**53**	304	57%
Sg, Med	0	26	1	**3**	2	**2**	**6**	2	**0**	**9**	0	**1**	**0**	52	40%
Sg, Dense	2	36	0	**2**	0	**4**	**3**	2	**1**	**5**	0	**3**	**15**	73	45%
Total	66	281	33	87	77	57	44	103	24	83	26	26	93	1,000	
Agreement															
RS to LEK	42%	53%	48%	59%	55%	88%	98%	57%	96%	100%	85%	69%	73%		
Overall															62%

Note: Here "RS to LEK agreement" is the probability that the RS map included habitats mapped by the LEK map. "LEK to RS agreement" is the probability that the LEK map included habitats mapped by the RS map. Overall agreement is the percentage of the map that was mapped the same with both the LEK and RS approaches. Bold numbers indicate matches. For (b) we considered habitat classes with the same habitats to match, since there was not a 1:1 relationship between habitat classes from the two mapping approaches. Co = Coral, Ru = Rubble, Sa = Sand, Sg = Seagrass, Ag = Algae

Appendix H Supporting materials for Chapter 5

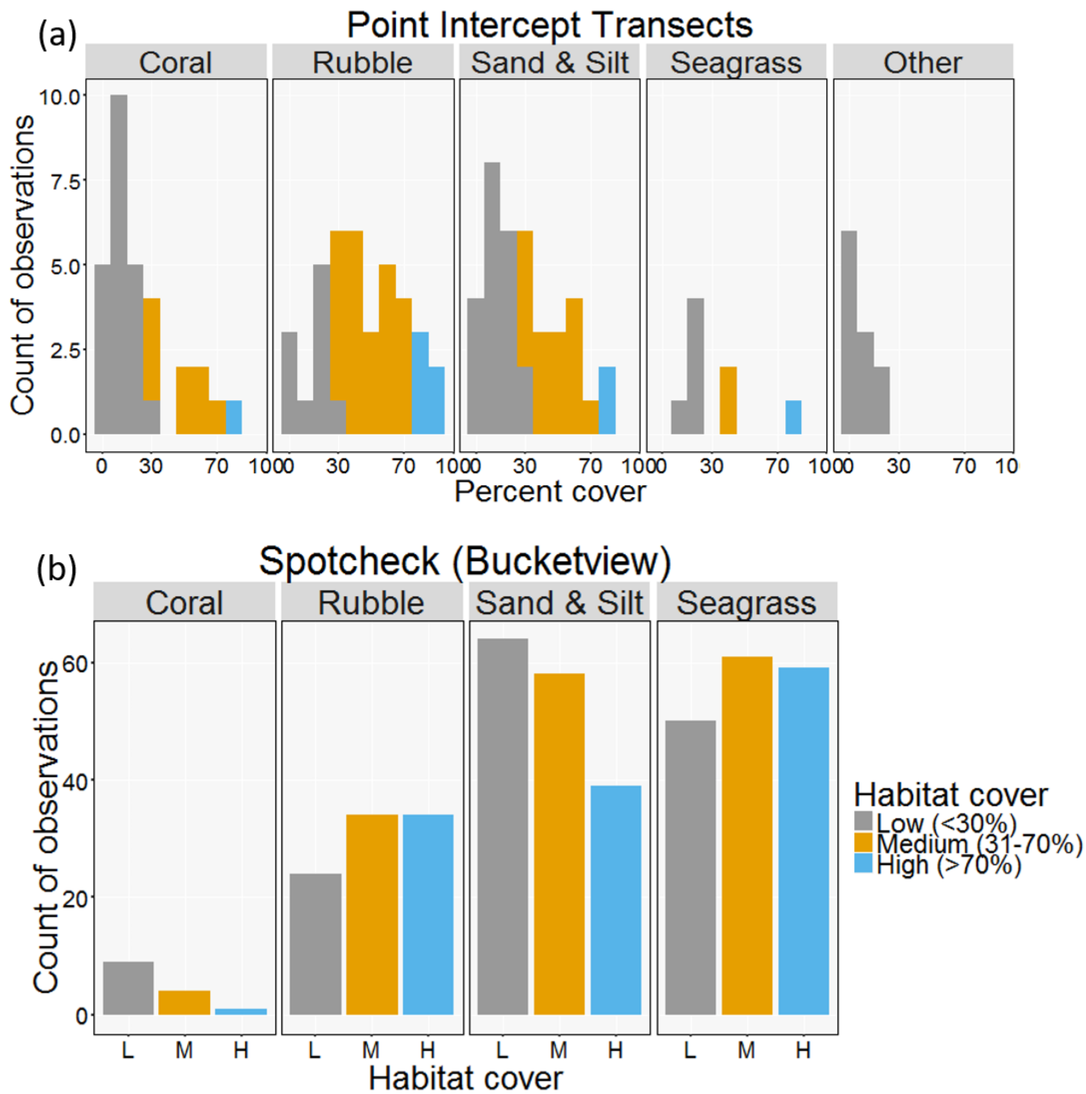

Figure H.1 The distribution of habitat cover data used for training image classification was collected using two georeferenced methods: (a) point intercept transects, and (b) spot check (bucketview). Data shown here are from surveys conducted in the study area in the central Danajon Bank, Philippines.

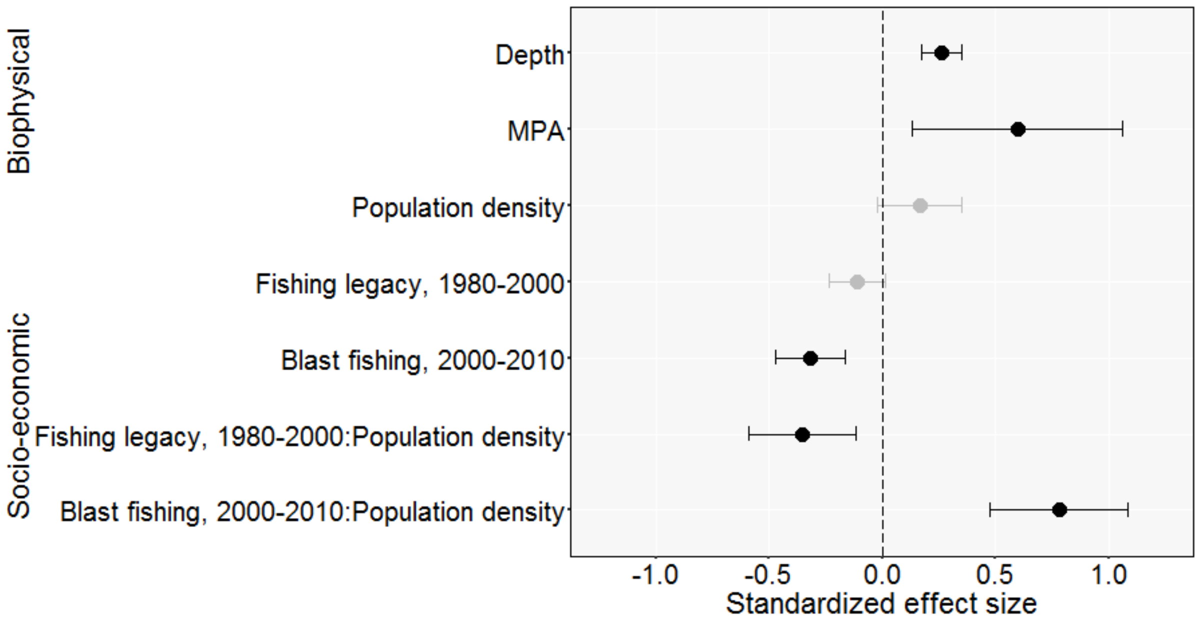

Figure H.2 Standardized effect sizes for reduced model predicting the probability of an area supporting living corals being present at locations throughout the Danajon Bank, Philippines. Reduced model included anthropogenic drivers and depth, but excluded seascape variables. Parameter estimates are from hierarchical logistic regression model with 95% confidence intervals. Grey points are not significant in model, but included for comparison with full model.